

**LITERATURE REVIEW OF THE WATER QUALITY EFFECTS
OF TIRE DERIVED AGGREGATE AND
RUBBER MODIFIED ASPHALT PAVEMENT**

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INTRODUCTION

Lack of easily accessible information on the water quality and environmental toxicity effects of tire derived aggregate (TDA) and rubber modified asphalt (RMA) was identified as by the U.S. Environmental Protection Agency Resource Conservation Challenge - Tire Cluster, Civil Engineering Subcommittee as an important barrier to expanding use of TDA in civil engineering applications. There have also been questions regarding runoff from rubber modified asphalt (RMA) pavement.

Studies are available that largely address the environmental concerns of using TDA and RMA, however, they have been published in a variety of journals and conference proceedings. Some results are only available in limited-distribution technical reports. Thus, when state environmental regulators are asked to approve use of TDA and RMA in their jurisdictions, they are sometimes hesitant because of the apparent lack of supporting data. Moreover, there are some studies from the late 1980's and early 1990's that had a limited scope and left unanswered questions. These older studies continue to be used as reasons to restrict TDA and RMA use, while ignoring the more complete data available in recent studies.

The purpose of this report is to present a review of literature on the water quality and environmental toxicology effects of TDA and RMA. The first part of this review focuses on field studies of TDA used both above and below the water table. This will be followed by a discussion of the water quality considerations of four specific applications: TDA used as fill for roadway, wall, building, and railroad construction; TDA used in landfill applications; TDA used as drain-field aggregate for septic tanks; and RMA pavements. Additional application specific data will be introduced as part of this discussion. A comprehensive list of references and an annotated bibliography is included as appendices.

FIELD STUDIES OF TDA USED ABOVE THE GROUNDWATER TABLE

There have been seven field studies that monitored the water quality effects of TDA placed above the groundwater table. These studies may be separated into two groups based on the methods used to collect the samples for analyses. In the first group, samples were taken directly from water that had passed through the TDA. With this method there is minimal opportunity for substances released by the TDA to be adsorbed by surrounding soil or diluted by mixing with groundwater. Projects in this category include: University of Wisconsin field trial (Eldin and Senouci, 1992; Bosscher, et al., 1993); North Yarmouth MaineDOT field trial (Humphrey, et al., 1997; Humphrey and Katz, 2000; Exponent, 2003; Sheehan, et al., 2006); Witter Farm Road field trail (Humphrey, 1999); Ohio tire shred monofills (Chyi, 2000); and Binghamton (New York) test embankment (Brophy and Graney, 2004). In the second group, samples were taken from groundwater monitoring wells located adjacent to the TDA fill. With this method substances released by the TDA may be adsorbed by the surrounding soil or diluted by mixing with groundwater. In addition, there may be a time delay between release of a substance by the TDA and its detection in the monitoring well. Projects in this category

includes: Dingley Road test project (Humphrey and Katz, 1995); Virginia DOT test embankment (Hoppe, 1998; Hoppe and Mullen, 2004); and Binghamton (New York) test embankment (Brophy and Graney, 2004).

The parameters selected for monitoring at each of the projects varied. All of the projects listed above were tested for a selection of metals and water quality index parameters such as pH. A more limited set of projects were tested for organic compounds (Humphrey, 1999; Humphrey and Katz, 2000; Exponent, 2003; Hoppe and Mullen, 2004; Sheehan, et al., 2006). On the North Yarmouth project the toxicity effects on aquatic life were examined (Exponent, 2003; Sheehan, et al., 2006). Only a few projects had an adequate control section where samples were taken to separate the water quality effects caused by the TDA from those that arise from naturally occurring substances at the site. This is important for the metals of interest since most are naturally occurring in the environment or may be present in the soil cover placed over the TDA. Moreover, it is possible that some organic compounds may be introduced into the project by sources that are unrelated to TDA including gasoline spills, oil leakage from vehicles, and air pollution. The durations of the studies varied from a single sample event to quarterly samples taken over a 5½-year period.

The remainder of this section organized as follows. First, the characteristics of each field site are summarized. Then, the statistical methods to used summarize the data are described followed by a statistical summary of the data. The summary is separated into sections on water quality index, inorganic chemicals with primary drinking water standards, inorganic chemicals with secondary drinking water standards, organic results, and toxicity results. The final section is a summary of the water quality effects of TDA placed above the water table.

Field Study Descriptions

University of Wisconsin Field Trial

An experimental test embankment was built as part of an access road at a landfill near Madison, Wisconsin. During construction pan lysimeters were placed beneath areas containing TDA to collect samples. One lysimeter was overlain by 5 ft (1.52 m) of TDA with a 4-in. (100-mm) nominal size topped with a 1-ft (0.3-m) thick soil cover. The other lysimeter was overlain by 5 ft (1.52 m) of TDA with a 2-in. (50-mm) nominal size topped with a 1-ft (0.3-m) thick soil cover. Each lysimeter was made of a 10-mil-thick PVC liner, and was covered with a nonwoven geotextile. Samples were tested 10 times over a 26 month period for eight metals, two anions, and five water quality index parameters (Eldin and Senouci, 1992; Bosscher, et al., 1993).

North Yarmouth MaineDOT Field Trial

This field trial was built on Route 231 in North Yarmouth, Maine, in the summer of 1993. It was studied from January 1994 to June 1999 by Humphrey and Katz (2000) to determine long-term water quality effects of TDA placed above the water table. It was subsequently sampled in 2000 and 2002 by Exponent (2003). The site consisted of five

100-ft (33-m) long sections, four of which were constructed using a 2-ft (0.61-m) thick layer of TDA topped with 2.5 to 4.5 ft (0.76 to 1.37 m) of granular soil and 0.42 ft (0.127 m) of pavement. The fifth section was established as a control section with conventional soil fill. About 100,000 tires processed to a maximum size of 3 in. (75 mm) were used. Two high-density polyethylene-lined seepage collection basins were installed under the sections containing TDA, and another basin was installed under the control area. The basins extended beyond the edge of the overlying pavement to capture runoff from the road that infiltrated into the road shoulder. Organic substances, inorganic substances with primary and secondary drinking water standards, pH, and other water quality index parameters were tested for by Humphrey and Katz (2000) and Exponent (2003). The samples taken by Exponent were also used to evaluate the toxicity effects on aquatic life (Exponent, 2003; Sheehan, et al., 2006).

Witter Farm Road Field Trial

A field trial was conducted to analyze insulation and drainage properties of a TDA layer placed below a paved road, as well as to collect data on the effects of TDA on water quality. The Witter Farm Road TDA field trial was constructed near the University of Maine in Orono, Maine using TDA with a maximum size of 3 in. (76 mm). The TDA or mixtures of TDA and aggregate base extended across the width of the road in a 6-in. (150-mm) or 12-in. (300-mm) thick layer. The TDA was topped by 13 in. (330 mm) or 25 in. (635 mm) of aggregate base followed by 5 in. (127 mm) of hot-mix asphalt pavement. In addition, a 6.5-ft (2.0-m) deep edge drain, backfilled with TDA or mixtures of TDA and aggregate base, was located beneath one shoulder. Water discharging from the edge drain was sampled on one date and analyzed for twelve metals, two anions, and organic compounds (Humphrey, 1999).

Ohio Tire Shred Monofills

In this study, leachate was collected from two Stark County, Ohio monofills (Chyi, 2000). The monofills contain tire shreds and soil cover. They are located in abandoned coal strip mines. The monofills have leachate collection systems so the results are applicable to above groundwater table conditions. The size and other characteristics of the tire shreds were not provided. The tire shred layer was 18 ft (5.5 m) thick at the C & E Coal site. The thickness was not given at the American Landfill site. Some of the samples were taken directly from the leachate collected from the bottom of the landfill. Samples were taken on eight dates spanning a two year period. The results are for filtered samples. The samples were analyzed for twelve commonly occurring cations and anions, as well as, an additional 19 trace elements. Results for total dissolved solids, pH, conductivity, and alkalinity were also reported. Selected results are compared to area groundwater data.

Dingley Road Test Project

This project consists of a 950-ft (290-m) long test site on Dingley Road in Richmond, Maine. TDA layer was 6 or 12 in. (150 or 300 mm) thick and the granular

soil cover was 12, 18, or 24 in. (300, 457, or 600 mm) thick. The project was constructed in August, 2002. The road was gravel surfaced at the time of water quality testing. About 20,000 tires were shredded to a maximum size of 2 in. (51 mm) for use on this project. Six PVC wells, including one positioned at a control site, were installed to collect groundwater samples. Water samples were taken on three occasions from December 1993 to January 1995 (Humphrey and Katz, 1995).

Virginia DOT Test Embankment

A mixture of TDA and soil was used to construct the approach fills for a new overpass for the Route 646 Connector located north of Williamsburg, Virginia. Up to 20 ft (6 m) of a mixture of 50/50 TDA/soil was placed. The soil used for mixing was silty sand. The section was topped by 5 ft (1.5 m) of soil. Two groundwater monitoring wells were installed one was located 500 ft (150 m) upgradient from the TDA/soil mixture section (MW-1), while the other (MW-2) was located immediately downgradient from the toe of the TDA/soil mixture section. The wells were monitored eight times between February, 1994 and January, 1997 for selected metals, hardness, pH, total organic carbon, total organic halides, and specific conductivity (Hoppe, 1999; Hoppe and Mullen, 2004).

Binghamton Test Embankment

TDA was used in construction of the subgrade for an interstate exit ramp near Binghamton, New York. The TDA layer was up to 10 ft (3 m) thick and was covered by 4.9 to 6.6 ft (1.5 to 2.0 m) of embankment fill (Dickson, et al., 2001). TDA had a nominal maximum size of 12 in. (300 mm). Three groundwater monitoring wells and two TDA fill sampling ports were installed. One of the monitoring wells was placed upgradient from the TDA, while the other two wells were positioned downgradient. One of the sampling ports was placed directly below the TDA fill area, and the other port was placed adjacent to the TDA fill area. Samples were monitored for selected metals and water quality index tests.

Minnesota Test Road

A study conducted for the Minnesota Pollution Control Agency (Twin City Testing, 1990) included limited field testing of TDA placed above the water table at two sites. Samples were taken from an open borehole at one site. There was a control well for this site, but it was located ¼-mile from the TDA site and the well was only one-third the depth so there is no assurance that background groundwater conditions were the same. Collecting water in this way is unlikely to produce representative samples. Therefore, this study will not be discussed further.

Data Summary

Statistical Methods

To allow for easy comparison of the data from the projects, the mean and standard deviation of each measured parameter was computed for each sample location that had three or more sampling events. For data sets where all the results were above the detection limit, the mean and standard deviation were computed using standard statistical methods. However, statistical analysis of data sets that have one or more results below the test method detection limit presents special challenges. In the past, several methods have been used including the following:

- Using only data that is above the detection limit. This results in a mean that is too high.
- Substituting the test method detection limit for all data points that are reported as nondetect. This also results in a mean that is too high.
- Substituting zero for all data points that are reported as nondetect. This results in a mean that is too low.
- Substituting half of the test method detection limit for all data points that are reported as nondetect. This may result in a mean that is either too high or too low.

The choice amongst the options listed above is arbitrary and none are supported by statistically sound reasoning (Helsel, 2005). Developing and promoting statistically sound methods to deal with analysis of water quality data containing nondetects has been the focus of work by Dr. Dennis R. Helsel of the U.S. Geological Survey. Helsel (2005) recommends the approach given below.

- Data sets with less than 50% nondetects. The mean and standard deviation should be computed using the Kaplan-Meier method which does not depend on the assumption of a distributional shape.
- Data sets with less than 50 data points and with 50 to 80% nondetects. Use a robust version of the “regression on order statistics” (ROS) method. In this method, the log of the data is plotted on a probability plot and a best fit straight line is plotted through the data. The line is used to obtain an estimate of the value of each data point that was nondetect. These values are then used along with the data above the detection limit to compute the mean and standard deviation using standard statistical methods.

- Data sets with greater than 80% nondetects. No meaningful statistics may be computed and only the percent of samples above the detection limit can be reported

The recommendations of Helsel (2005) were used to analyze the data in this summary.

Projects with Direct Collection of Samples

The following five projects collected water samples directly from TDA placed above the water table: University of Wisconsin field trial (Eldin and Senouci, 1992; Bosscher, et al., 1993); North Yarmouth MaineDOT field trial (Humphrey, et al., 1997; Humphrey and Katz, 2000; Exponent, 2003; Sheehan, et al., 2006); Witter Farm Road field trail (Humphrey, 1999); Ohio tire shred monofills (Chyi, 2000); and Binghamton (New York) test embankment (Brophy and Graney, 2004). The results for water quality index tests, inorganic chemicals, organic chemicals, and aquatic toxicity tests are summarized in the following sections.

Water Quality Index Tests

A range of water quality index tests were performed in the field studies including: pH, total dissolved solids (TDS), total solids (TS), biological oxygen demand (BOD), and chemical oxygen demand (COD). The mean results are summarized in Table 1. The mean pH from each of the projects varied from 6.79 to 7.51 indicating that the conditions were near neutral and within secondary drinking water standard (USEPA, 2006).

For the North Yarmouth project hypothesis testing (Neter, et al., 1982) showed that the TDS, BOD, and COD in TDA sections C and D were statistically equal to the control section with a confidence level of 90%. This indicates that the presence of TDA did not affect these parameters.

The TDS for the North Yarmouth project for both the control and the two TDA sections exceeded the secondary drinking water standard, as well as, the Wisconsin West Lysimeter exceeded that secondary standard. However, based on the statistical comparison discussed in the previous paragraph this is not likely due to the TDA.

Table 1. Mean values of water quality index tests from field studies with direct collection of samples.

Analyte	Secondary Standard	Wisconsin		North Yarmouth			Witter Farm Road**	Ohio Monofills		Binghamton, NY	
		West 4" TDA	East 2" TDA	Control	TDA Section C	TDA Section D		C&E	American	Control TF2	TDA TF1
pH	6.5-8.5	7.5	7.5	7.21	7.11	7.05	NA	7.29	7.51	7.06	6.79
Total dissolved solids (mg/L)	500	1883	230	1096	1049	902	320	NA	NA	NA	NA
Total solids (mg/L)	NA	NA	NA	1117	1062	925	610	NA	NA	NA	NA
BOD(mg/L)	NA	16.3	38.7	1.79	1.34	2.04	NA	NA	NA	NA	NA
COD (mg/L)	NA	163	279	56.6	58.0	61.1	NA	NA	NA	NA	NA

NA = not available – parameter not tested for

* Results for unfiltered sample reported

**Results from a single sample reported

References: Wisconsin (Edil and Bosscher, 1992; Eldin and Senouci, 1992; Bosscher, et al., 1993); North Yarmouth (Humphrey and Katz, 2000); Witter Farm Road (Humphrey, 1999); Ohio Monofills (Chyi, 2000); Binghamton (Brophy and Graney, 2004); Secondary standard (USEPA, 2006)

Inorganic Chemicals with a Primary Drinking Water Standard

Thirteen inorganic chemicals with primary drinking water standards were directly monitored on filtered samples in one or more of the field studies with the exception of mercury (Hg), where only results for unfiltered samples were available. Since wells for drinking water are designed to minimize infiltration of solids, it is appropriate to compare drinking water standards to the results from filtered samples when available. No studies considered the following chemicals with primary drinking water standards: asbestos, cyanide, and nitrite; however, there is no reason to believe that these inorganic chemicals are present in tires. With these exceptions all of the inorganic chemicals with primary drinking water standards were considered in one or more studies.

The mean concentrations and regulatory allowable limits (RAL) are shown in Table 2. The mean concentrations were less than the RAL for the Wisconsin, North Yarmouth, Witter Farm Road, American Monofill, and Binghamton projects. In contrast, the C&E Monofill exhibited concentrations of antimony (Sb), arsenic (As), lead (Pb), selenium (Se), and thallium (Ti) that were above their corresponding RAL. No control samples were taken from this project, so it is not possible to determine if the source of these substances was naturally occurring groundwater, the soil used to cover the tire shreds, or the tire shreds themselves. The other projects that includes these analytes, found that the concentrations were below the RAL. Moreover, these substances are not used in the manufacture of tires.

For the two projects with control sections (North Yarmouth and Binghamton), hypothesis testing (Neter, et al., 1982) was performed to evaluate the relative magnitudes of the mean concentrations from the control and TDA sections. For the analytes where it was possible to compute a mean for both control and TDA sections (barium and chromium at North Yarmouth, and barium and cadmium at Binghamton), it was found that the mean concentrations in the sections with TDA were equal to or less than the concentration in the corresponding control section when evaluated with a confidence level of 90%. For three additional analytes at the North Yarmouth site (cadmium, copper, and lead), the percent of samples below the test method detection limit for the TDA sections was greater than or equal to that of the control sections.

In summary, the preponderance of evidence shows that TDA would not cause primary drinking water standards to be exceeded. Moreover, the evidence shows that TDA is unlikely to increase levels of metals with primary drinking water standards above naturally occurring background levels.

Table 2. Mean concentrations of inorganic analytes with primary drinking water standards from field studies with direct collection of samples.

Analyte	RAL	Wisconsin		North Yarmouth			Witter Farm Road**	Ohio Monofills		Binghamton, NY	
		West 4''TDA	East 2''TDA	Control	TDA Section C	TDA Section D		C&E Mono-fill	American Monofill	Contro 1 TF2	TDA TF1
antimony (Sb)	0.006	NA	NA	100%<0.05*	100%<0.05*		NA	0.1290	100%<0.005	NA	NA
arsenic (As)	0.010	NA	NA	NA	NA	NA	NA	0.31	67%<0.001	NA	NA
barium (Ba)	2.0	0.346	0.281	0.0688	0.0339	0.0395	0.017	0.218	0.0603	0.796	0.392
beryllium (Be)	0.004	NA	NA	100%<0.005*	100%<0.005*		NA	100%<0.1	100%<0.001	NA	NA
cadmium (Cd)	0.005	NA	NA	95%<0.0005	100%<0.0005	96%<0.0005	<0.0005	80%<0.1	67%<0.001	0.0325	0.00867
chromium (Cr)	0.1	NA	NA	0.0118	0.0126	0.0119	<0.006	NA	NA	NA	NA
copper (Cu)	1.3	NA	NA	91%<0.009	91%<0.009	96%<0.009	<0.009	80%<0.02	67%<0.01	NA	NA
fluoride (F)	4.0	NA	NA	NA	NA	NA	NA	0.8018	0.7356	NA	NA
lead (Pb)	0.015	90%<0.003	0.008	88%<0.002	88%<0.002	94%<0.002	<0.002	0.19	67%<0.001	NA	NA
mercury (Hg)	0.002	NA	NA	100%<0.0005*	100%<0.0005*		NA	NA	NA	NA	NA
nitrate (NO ₃ ⁻)	10	NA	NA	NA	NA	NA	NA	0.9217	0.8933	NA	NA
selenium (Se)	0.05	NA	NA	NA	NA	NA	NA	0.231	100%<0.001	NA	NA
thallium (Ti)	0.002	NA	NA	NA	NA	NA	NA	80%<0.002	100%<0.002	NA	NA

Units = mg/l

When possible, the calculated mean is reported; if the mean could not be calculated because of limited number of samples with concentrations above the detection limit, then the percent of the results below the detection limit is reported.

NA = not available – parameter not tested for

*Results from two unfiltered samples reported by Exponent (2003); results for TDA are from a composite sample of TDA sections C and D

**Results from a single sample reported

References: Wisconsin (Eldin and Senouci, 1992; Bosscher, et al., 1993); North Yarmouth (Humphrey and Katz, 2000; Exponent, 2003); Witter Farm Road (Humphrey, 1999); Ohio Monofills (Chyi, 2000); Binghamton (Brophy and Graney, 2004); RAL (USEPA, 2006)

Inorganic Substances with a Secondary Drinking Water Standard

Nine inorganic chemicals with secondary drinking water standards were directly monitored on filtered samples in one or more of the field studies. As noted previously, wells for drinking water are designed to minimize infiltration of solids, so it is appropriate to compare drinking water standards to the results from filtered samples when available. Secondary standards are “non-enforceable guidelines regulating contaminants that may cause cosmetic effects (such as skin or tooth discoloration) or aesthetic effects (such as taste, odor, or color) in drinking water” (USEPA, 2006). In addition to the nine inorganic chemicals discussed in this section, secondary standards for pH and total dissolved solids were discussed previously in the section on water quality index tests.

The mean concentrations and secondary standards are shown in Table 3. The mean concentrations of aluminum, copper, fluoride, silver, and zinc were below their corresponding secondary standard except for aluminum at the C&E monofill. No control samples were taken from this project, so it is not possible to determine if the source of the aluminum was naturally occurring in the groundwater, the soil used to cover the tire shreds, or the tire shreds themselves. This notwithstanding, the preponderance of evidence indicates that TDA will not cause the secondary standard for aluminum to be exceeded.

TDA causes elevated levels of iron (Fe) and manganese (Mn). This can be seen by comparing the control and TDA sections at North Yarmouth and Binghamton. Moreover, the concentrations are above their corresponding secondary standards.

The mean chloride concentrations exceeded its secondary standard for the Wisconsin and North Yarmouth projects. Both of these projects are located in cold climates where road salt is routinely applied as part of winter road maintenance. This is the likely source of the chloride and this analyte will not be discussed further.

The mean concentrations for sulfate were less than its secondary standard with the exception of the two Ohio monofill projects. This compound is associated with acid mine drainage from coal mines, thus construction of these monofills in abandoned coal mines likely is the cause of the elevated levels of sulfate.

For the two projects with control sections (North Yarmouth and Binghamton), hypothesis testing (Neter, et al., 1982) was performed to evaluate the relative magnitudes of the mean concentrations from the control and TDA sections. For North Yarmouth project, the zinc and sulfate concentration in the TDA sections was less than the control section when evaluated with a confidence level of 90%. Likewise, at the Binghamton site, the zinc concentration in the TDA section was less than in the control with a significance of 90%. Sulfate was not monitored at the Binghamton project. In contrast, for both projects the iron and manganese concentrations in the TDA was greater than the control with a significance of 90%. For two additional analytes at the North Yarmouth site (aluminum and copper), the percent of samples below the test method detection limit for the TDA sections was greater than or equal to that of the control sections.

Table 3. Mean concentrations of inorganic analytes with secondary drinking water standards from field studies with direct collection of samples.

Analyte	Secondary Standard	Wisconsin		North Yarmouth			Witter Farm Road**	Ohio Monofills		Binghamton, NY	
		West 4" TDA	East 2" TDA	Control	TDA Section C	TDA Section D		C&E	American	Control TF2	TDA TF1
aluminum (Al)	0.2	NA	NA	81%<0.07	100%<0.07	100%<0.07	<0.07	7.97	67%< 0.1	NA	NA
chloride (Cl ⁻)	250	477	600	345.8*	331.9*	338.*	111	44.2	34.6	NA	NA
copper (Cu)	1	NA	NA	91%<0.009	91%<0.009	96%<0.009	<0.009	80%<0.02	67%<0.01	NA	NA
fluoride (F)	2.0	NA	NA	NA	NA	NA	NA	0.80	0.736	NA	NA
iron (Fe)	0.3	0.71	1.13	0.0198	0.0795	0.555	0.158	0.19	0.103	0.255	15.0
manganese (Mn)	0.05	1.129	1.522	0.0421	4.38	2.56	2.53	2.72	1.93	0.260	6.21
silver (Ag)	0.10	NA	NA	NA	NA	NA	NA	80%<0.005	100%<0.001	NA	NA
sulfate (SO ₄ ²⁻)	250	115	213	25.3*	18.9*	11.4*	3.51	468.5	600.7	NA	NA
zinc (Zn)	5	0.093	0.230	1.10	0.0111	0.0111	0.082	0.492	100%<0.005	0.300	0.0343

Units = mg/l

When possible, the calculated mean is reported; if the mean could not be calculated because of limited number of samples with concentrations above the detection limit, then the percent of the results below the detection limit is reported.

NA = not available – parameter not tested for

* Results for unfiltered sample reported

**Results from a single sample reported

References: Wisconsin (Eldin and Senouci, 1992; Bosscher, et al., 1993); North Yarmouth (Humphrey and Katz, 2000); Witter Farm Road (Humphrey, 1999); Ohio Monofills (Chyi, 2000); Binghamton (Brophy and Graney, 2004); Secondary standard (USEPA, 2006)

Organic Compounds

There are two field studies that monitored for organics in samples taken directly from water that had passed through the TDA. The projects are: North Yarmouth MaineDOT field trial (Humphrey, et al., 1997; Humphrey and Katz, 2000; Exponent, 2003; Sheehan, et al., 2006); and Witter Farm Road field trail (Humphrey, 1999). The North Yarmouth Project was tested for organics five times: 12/28/1995, 04/05/1996, 06/22/1999, 11/08/2000, and 01/01/2002. The Witter Farm Road project was sampled on a single date (06/27/97).

Samples were tested for volatile organic compounds (VOCs) in accordance with EPA Method 8260 (Determination of Volatile Organics by Purge-and-Trap Capillary Column GC/MS). For the first three sample dates at North Yarmouth and the single sample date at Witter Farm Road, 82 VOCs were targeted for analysis (Humphrey and Katz, 2000; Humphrey, 1999). For the last two sample dates at North Yarmouth 66 VOCs were targeted for analysis (Exponent, 2003). In addition, the test method reporting limit was generally lower for these dates.

Samples were tested for semivolatile organic compounds (SVOCs) in accordance with EPA Method 8270 (Determination of Semivolatile Organics by Capillary Column GC/MS). For the first three sample dates at North Yarmouth and the single sample date at Witter Farm Road, 69 SVOCs were targeted for analysis (Humphrey and Katz, 2000; Humphrey, 1999). For the last two sample dates at North Yarmouth 68 SVOCs were targeted for analysis (Exponent, 2003). In addition, the test method reporting limit was generally lower for these dates.

Results for the ten compounds that were above the test method reporting limit in one or more samples from the North Yarmouth project are summarized in Table 4. Regulatory allowable limits (RAL) have been established for three of the compounds. The detected concentrations were below the RAL. Four of the occurrences with concentrations above the detection limit were in the control section. Moreover, the highest single concentration of a compound (0.100 mg/L of 3&4-methylphenol) was found in the control section. The source of these compounds is unknown, but the control section was located uphill from the TDA sections so it could not have been due to the TDA. In the TDA sections, there were five instances of compounds being found above the test method reporting limit, with the highest concentration being 0.024 mg/L of acetone on 1/1/2002. In addition, there were two instances of compounds being detected at concentrations below the test method reporting limit.

The results for the single sample from the Witter Farm Road were all below the test method reporting limit.

Table 4 – Concentrations on organic analytes from North Yarmouth project.

North Yarmouth - Control							
Compound		RAL	12/28/1995	4/5/1996	6/22/1999	11/8/2000	1/1/2002
1,1-dichloroethane	VOC	0.005	ND*	ND*	ND*	ND [#]	ND [#]
cis-1,2-dichloroethene (1)	VOC	0.070	ND*	ND*	ND*	ND [#]	ND [#]
toluene	VOC	1.000	ND*	ND*	0.070	ND [#]	ND [#]
4-methyl-2-pentanone	VOC	----	ND*	ND*	ND*	ND**	ND**
acetone	VOC	----	ND**	ND**	ND**	ND**	ND**
aniline	SVOC	----	ND**	ND**	ND**	ND*	ND*
bis(2-ethylhexyl)phthalate	SVOC	----	ND**	ND**	ND**	ND*	0.0060
3&4-methylphenol	SVOC	----	ND**	ND**	0.100	ND*	ND*
benzoic acid	SVOC	----	ND**	ND**	0.025	ND*	ND*
phenol	SVOC	----	ND**	ND**	0.074	ND*	ND*
North Yarmouth – TDA Section C							
Compound		RAL	12/28/1995	4/5/1996	6/22/1999	11/8/2000	1/1/2002
1,1-dichloroethane	VOC	0.005	ND*	ND*	ND*	0.0013	ND [#]
cis-1,2-dichloroethene (1)	VOC	0.070	ND*	ND*	ND*	0.0015	ND [#]
toluene	VOC	1.000	ND*	ND*	ND*	ND [#]	ND [#]
4-methyl-2-pentanone	VOC	----	ND*	ND*	ND*	ND**	ND**
acetone	VOC	----	ND**	ND**	ND**	0.0140	0.0240
aniline	SVOC	----	ND**	ND**	ND**	0.0050	ND*
bis(2-ethylhexyl)phthalate	SVOC	----	ND**	ND**	ND**	ND*	ND*
3&4-methylphenol	SVOC	----	ND**	ND**	ND**	ND*	ND*
benzoic acid	SVOC	----	ND**	ND**	ND**	ND*	ND*
phenol	SVOC	----	ND**	ND**	ND**	ND*	ND*
North Yarmouth – TDA Section D							
Compound		RAL	12/28/1995	4/5/1996	6/22/1999	11/8/2000	1/1/2002
1,1-dichloroethane	VOC	0.005	ND*	ND*	<0.005	On 11/8/2000 and 1/1/2002 the sample was a composite from Sections C and D - see above for results	
cis-1,2-dichloroethene (1)	VOC	0.070	ND*	ND*	ND*		
toluene	VOC	1.000	ND*	ND*	ND*		
4-methyl-2-pentanone	VOC	----	ND*	ND*	<0.005		
acetone	VOC	----	ND**	ND**	ND**		
aniline	SVOC	----	ND**	ND**	ND**		
bis(2-ethylhexyl)phthalate	SVOC	----	ND**	ND**	ND**		
3&4-methylphenol	SVOC	----	ND**	ND**	ND**		
benzoic acid	SVOC	----	ND**	ND**	ND**		
phenol	SVOC	----	ND**	ND**	ND**		

FOOTNOTES

[#]MRL=0.0005^{*}MRL=0.005^{**}MRL=0.010

(1) cis-1,2-dichloroethene also known as cis-1,2-dichloroethylene

Aquatic Toxicity

Samples from the North Yarmouth Project were used to perform two USEPA (USEPA, 1991) freshwater short-term toxicity tests. One test was three-brood survival and reproduction test with the crustacean *Ceriodaphnia dubia*. In this test, females are exposed to samples for the period of time it takes for control treatment females to produce three broods, typically about 7 days. Then survival and reproduction of these two groups are compared. The other test was seven-day survival and growth of larval fathead minnows (*Pimephales promelas*). At the completion of the test, survival and growth for minnows exposed to field samples are compared to control samples. Samples were taken on two dates: 11/8/2000 and 1/1/2002. On each date samples were taken from the control section and a composite of TDA Sections C+D.

On both sample dates there was 100% survival and growth of fathead minnows in samples from the control section and TDA Sections C+D. For *Ceriodaphnia dubia* there was 100% survival and reproduction in the samples from TDA Sections C+D on both sample dates. However for the control section, there was 100% survival and 49% reproduction on the first sample date. On the second sample date the results in the control section had decreased to 49% survival and 28% reproduction. These toxicity effects were confined to the control section and could not have been a result of the TDA. These results show TDA placed above the groundwater table on this project had no effect on aquatic toxicity for these two species.

Projects with Samples from Adjacent Wells

Three projects had samples taken from wells that were adjacent to the project: Dingley Road in Richmond, Maine; Binghamton Test Embankment in Binghamton, New York; and the Virginia DOT Test Embankment located north of Williamsburg, Virginia. Each of these projects had a control well that was located upgradient of the TDA section, and one or more wells located adjacent to the TDA section. As noted previously, with this sampling method substances released by the TDA may be absorbed by the surrounding soil or diluted by mixing with groundwater. Moreover, there may be a time delay between release of a substance by the TDA and its detection in the monitoring well.

Water Quality Index Tests

A range of water quality index tests were performed on projects with samples from adjacent wells including: pH, total dissolved solids (TDS), total solids (TS), biological oxygen demand (BOD), and chemical oxygen demand (COD). The mean results are summarized in Table 5. The pH from the Richmond, Maine project was within the secondary drinking water standard (USEPA, 2006). However, two wells from the Binghamton project and both wells from the Virginia project were more acidic than the standard. For the Binghamton and Virginia projects the pH was slightly lower in the wells adjacent to the TDA sections compared to the control wells. However, the pH for the control well at the Richmond project was in the middle of the range of the five wells adjacent to the TDA sections. The TDS was below the secondary drinking water standard for the Richmond project, which was the only project that included this parameter.

Table 5. Mean values of water quality index tests from field studies with indirect collection of samples.

Analyte	Secondary Standard	Dingley Road, Richmond, Maine						Binghamton, NY			Virginia	
		0+69 (control)	3+00 (TDA)	3+42 (TDA)	6+19 (TDA)	6+77 (TDA)	8+32 (TDA)	FH-X-138 (control)	FH-X-142 (TDA)	FH-X-143 (TDA)	Well #1 (control)	Well #2 (TDA)
pH	6.5-8.5	7.04	6.87	6.84	7.57	7.51	6.90	7.09	6.20	6.18	4.82	4.30
TDS(mg/L)	500 mg/L	430*	210*	NA	58*	49*	278*	NA	NA	NA	NA	NA
TS (mg/L)	----	264*	97*	NA	61*	49*	295*	NA	NA	NA	NA	NA
BOD (mg/L)	----	6.38	3.99	4.25	3.46	4.83	1.03	NA	NA	NA	NA	NA
COD (mg/L)	----	66.7%<10	66.7%<10	50%<10	66.7%<10	66.7%<10	66.7%<10	NA	NA	NA	NA	NA

When possible, the calculated mean is reported; if the mean could not be calculated because of limited number of samples with concentrations above the detection limit, then the percent of the results below the detection limit is reported.

NA = not available – parameter not tested for

* Results for a single unfiltered sample reported.

**Detection limit for lead (Pb) on Richmond project was 0.057 mg/L on two sampling dates and 0.005 mg/L on one sampling date.

***Detection limit for cadmium (Cd) on Binghamton project not reported.

References: Dingley Road (Humphrey and Katz, 1995); Binghamton, NY (Brophy and Graney, 2004); Virginia (Hoppe, 1999; Hoppe and Mullen, 2004); RAL (USEPA, 2006)

Inorganic Chemicals with a Primary Drinking Water Standard

Five inorganic chemicals with primary drinking water standards (barium, cadmium, chromium, copper, and lead) were monitored on filtered samples in one or more of the field studies. In addition, on one date unfiltered samples were tested for selenium on one project. The mean concentrations and regulatory allowable limits (RAL) are shown in Table 6. The barium concentration in the control well at the Binghamton Project was above the RAL. The reason is unknown, but it could not be due to the TDA since this well is located upgradient of the TDA fill. With this exception, the results in both the control and TDA wells were below the corresponding RAL. Further statistical comparison was not possible because the limited number of sample events combined with the high percentage of sample results that were below the detection limit.

Inorganic Substances with a Secondary Drinking Water Standard

Seven inorganic chemicals with secondary drinking water standards were monitored in one or more of the field studies with adjacent monitoring wells. As noted previously, wells for drinking water are designed to minimize infiltration of solids, so it is appropriate to compare drinking water standards to the results from filtered samples when available. Secondary standards are “non-enforceable guidelines regulating contaminants that may cause cosmetic effects (such as skin or tooth discoloration) or aesthetic effects (such as taste, odor, or color) in drinking water” (USEPA, 2006). In addition to the seven inorganic chemicals discussed in this section, secondary standards for pH and total dissolved solids were discussed previously in the section on water quality index tests.

The mean concentrations and secondary standards are shown in Table 7. The mean concentrations of aluminum, chloride, sulfate, and zinc were below their corresponding secondary standard on the projects where these parameters were measured. The secondary standard for iron was exceeded on the Binghamton and Virginia projects for both the control wells and the wells adjacent to the TDA sections. On the Binghamton project iron levels were higher in the wells adjacent to the TDA section compared to the control well. However, the iron concentration was about the same in the TDA and control wells in the Virginia project. The secondary drinking water standard for manganese was exceeded for all control and TDA wells. For the Richmond project, the manganese level in three TDA wells was about the same as the control well, while for the remaining three wells the concentration in the TDA wells was lower than the control well. For the Binghamton project, the manganese concentrations in the TDA wells were elevated. For the Virginia project, the control and TDA wells had about the same manganese concentrations.

Table 6. Mean concentrations of inorganic analytes with primary drinking water standards from field studies with indirect collection of samples.

Analyte	RAL	Dingley Road, Richmond, Maine						Binghamton, NY			Virginia	
		0+69 (control)	3+00 (TDA)	3+42 (TDA)	6+19 (TDA)	6+77 (TDA)	8+32 (TDA)	FH-X-138 (control)	FH-X-142 (TDA)	FH-X-143 (TDA)	Well #1 (control)	Well #2 (TDA)
barium (Ba)	2.0	33.3%<0.01	33.3%<0.01	50%<0.01	33.3%<0.01	66.7%<0.01	33.3%<0.01	13.43	0.290	0.299	NA	NA
cadmium (Cd)	0.005	100%<0.005	100%<0.005	100%<0.005	100%<0.005	100%<0.005	100%<0.005	100%<***	33.3%<***	0.0076	NA	NA
chromium (Cr)	0.1	0.013	66.7%<0.01	50%<0.01	66.7%<0.01	33.3%<0.01	33.3%<0.01	NA	NA	NA	NA	NA
copper (Cu)	1.3	33.3%<0.01	100%<0.01	100%<0.01	66.7%<0.01	100%<0.01	100%<0.01	NA	NA	NA	1.85	1.3
lead (Pb)	0.015	100%<DL**	100%<DL**	100%<DL**	100%<DL**	100%<DL**	100%<DL**	NA	NA	NA	0.006	0.007
selenium (Se)*	0.05	<0.001	<0.001	NA	<0.001	<0.001	<0.001	NA	NA	NA	NA	NA

Units = mg/l

When possible, the calculated mean is reported; if the mean could not be calculated because of limited number of samples with concentrations above the detection limit, then the percent of the results below the detection limit is reported.

NA = not available – parameter not tested for

* Selenium results for a single unfiltered sample reported.

**Detection limit for lead (Pb) on Richmond project was 0.057 mg/L on two sampling dates and 0.005 mg/L on one sampling date.

***Detection limit for cadmium (Cd) on Binghamton project not reported.

References: Dingley Road (Humphrey and Katz, 1995); Binghamton, NY (Brophy and Graney, 2004); Virginia (Hoppe, 1999; Hoppe and Mullen, 2004); RAL (USEPA, 2006)

Table 7. Mean concentrations of inorganic analytes with secondary drinking water standards from field studies with indirect collection of samples.

Analyte	Secondary Standard	Dingley Road, Richmond, Maine						Binghamton, NY			Virginia	
		0+69 (control)	3+00 (TDA)	3+42 (TDA)	6+19 (TDA)	6+77 (TDA)	8+32 (TDA)	FH-X-138 (control)	FH-X-142 (TDA)	FH-X-143 (TDA)	Well #1 (control)	Well #2 (TDA)
aluminum (Al)	0.2	100%<0.07	100%<0.07	50%<0.07	100%<0.07	100%<0.07	100%<0.07	NA	NA	NA	NA	NA
chloride (Cl ⁻)	250	111	103	9.2	4.3	2.7	141	NA	NA	NA	15.1	12.1
copper (Cu)	1	33.3%<0.01	100%<0.01	100%<0.01	66.7%<0.01	100%<0.01	100%<0.01	NA	NA	NA	1.85	1.3
iron (Fe)	0.3	100%<0.1	100%<0.1	50%<0.1	100%<0.1	100%<0.1	100%<0.1	0.41	7.06	1.29	0.47	0.45
manganese (Mn)	0.05	13.5	11.8	2.00	1.80	1.37	15.8	0.61	11.3	14.4	4.9	5.1
sulfate (SO ₄ ²⁻)	250	48.3	9.47	7.95	5.10	8.53	7.83	NA	NA	NA	NA	NA
zinc (Zn)	5	66.7%<0.01	100%<0.01	100%<0.01	100%<0.01	100%<0.01	100%<0.01	0.091	0.112	0.098	0.127	0.123

Units = mg/l

When possible, the calculated mean is reported; if the mean could not be calculated because of limited number of samples with concentrations above the detection limit, then the percent of the results below the detection limit is reported.

NA = not available – parameter not tested for

References: Dingley Road (Humphrey and Katz, 1995); Binghamton, NY (Brophy and Graney, 2004); Virginia (Hoppe, 1999; Hoppe and Mullen, 2004); Secondary Standard (USEPA, 2006)

Organic Compounds

The Virginia DOT project monitored total organic carbon (TOC) and total organic halides (TOX). For the control well, TOC was below the detection limit of 1 mg/L for 16 out of 32 tests. For the well adjacent to the TDA section, 28 out of 32 tests were below the detection limit. For the control well, TOX was below the detection limit of 0.02 mg/L for 24 out of 32 tests. For the well adjacent to the TDA section, 28 out of 32 tests were below the detection limit. Thus, there was no indication that the TDA increased the TOC and TOX in the groundwater.

Summary of Water Quality Effects of TDA Placed Above the Groundwater Table

The preponderance of evidence shows that TDA does not cause metals with primary drinking water standards to be exceeded. Moreover, the data shows that TDA is unlikely to increase levels of metals with primary drinking water standards above naturally occurring background levels.

For chemicals with secondary drinking water standards, it is likely that TDA would increase the concentrations of iron and manganese. For water that is collected as it emerges from a TDA fill, it is likely that the levels of these substances will exceed secondary drinking water standards. However, for two of three projects where samples were taken from wells adjacent to the TDA fills, the iron and manganese levels were about the same as background levels. The prevalence of naturally occurring manganese is shown by the concentrations in the control wells at the Richmond, Binghamton and Virginia projects which exceeded the secondary drinking water standard. For other chemicals with secondary drinking water standards, there is no evidence that TDA affects naturally occurring background levels.

Volatile and semivolatile organics were monitored on two projects. Substances are generally below test method reporting limits. Moreover, for those substances with drinking water standards, the levels were below the standards. A few substances are occasionally found above the test method detection limit, however, the highest concentrations were found in the control section indicating that organics from other sources associated with active roadways are more prevalent.

Aquatic toxicity tests were performed on samples taken from one project. The results showed that water collected directly from TDA fills had no effect on survival, growth, and reproduction of two standard test species (fathead minnows and a small crustacean, *Ceriodaphnia dubia*).

FIELD STUDIES OF TDA USED BELOW THE GROUNDWATER TABLE

The water quality effects of TDA placed below the groundwater table at three field sites was studied by Humphrey and Katz (2001). In this study, 1.5 tons (1.4 metric tons) of TDA was buried below the groundwater table in the following three soil types: peat (P), marine clay (C), and glacial till (T). The TDA was made from a mixture of steel and glass belted tires and had a maximum size of 3 in. (75 mm). The TDA was placed in

2 to 6-ft (0.7 to 1.8-m) wide trenches with their long axis oriented perpendicular to the approximate direction of groundwater flow. At each site, one monitoring well was installed up gradient of the TDA filled trench to obtain background water quality; one well was installed directly in the TDA filled trench; and one to three wells were installed 2 ft (0.6 m) down gradient of the trench; and one well was installed about 10 ft (3 m) down gradient of the trench. Samples were taken for a range of metals, volatile organics, and semivolatile organics (Humphrey and Katz, 2001). Aquatic toxicity tests were also performed (Exponent, 2003; Sheehan, et al., 2006). Samples were taken on seven dates between 1994 and 1997 by Humphrey and Katz (2001). Additional samples were taken at the peat site by Exponent (2003) in 2000 and 2001.

Metals with Primary Drinking Water Standards

The following metals with a primary drinking water standard were included in the study by Humphrey and Katz (2001): arsenic (As), barium (Ba), cadmium (Cd), chromium (Cr), copper (Cu), and lead (Pb). In addition, antimony (Sb), beryllium (Be), mercury (Hg), selenium (Se), and thallium (Tl) were included in the samples from the peat site tested by Exponent (2003). The concentration of arsenic (As), cadmium (Cd), and lead (Pb) were below the test method detection limit in all samples (15, 5, and 15 µg/L, respectively) tested by Humphrey and Katz (2001). Likewise, the concentration of cadmium (Cd) was below the test method detection limit (5 µg/L) in all samples at the peat site tested by Exponent (2003). However, Exponent (2003) found arsenic at concentrations up to 8 µg/L in the upgradient and downgradient wells, but was below the detection limit of 5 µg/L in the TDA filled trench. Given the absence in the TDA filled trench the arsenic must have been from natural sources. Lead (Pb) was found by Exponent (2003) at a concentration of 5 µg/L on one date in the TDA filled trench but this is below the regulatory allowable limit of 15 µg/L (USEPA, 2006).

The concentration of dissolved copper (Cu) was generally below the test method detection limit (3 µg/L for Humphrey and Katz, 2001; 10µg/L for Exponent, 2003), however, background levels in a few samples were as high as 11 µg/L compared to levels up to 4 µg/L in the TDA filled trenches. Thus, TDA does not appear to increase the levels of dissolved copper (Cu).

The concentration of dissolved barium (Ba) in the TDA filled trench ranged from 7 to 57 µg/L compared to background levels of 6 to 33 µg/L. In the wells located 2 ft (0.6 m) downgradient of the TDA filled trench the concentration of barium (Ba) had returned to background levels (5 to 39 µg/L). Thus, TDA submerged in groundwater may slightly increase the level of barium (Ba), but Ba did not show a tendency to migrate downgradient. More importantly, the level of dissolved Ba was less than its primary drinking water standard of 2000 µg/L, even in the TDA filled trenches (Humphrey and Katz, 2001; Exponent, 2003).

The concentrations of antimony (Sb), beryllium (Be), mercury (Hg), selenium (Se), and thallium (Tl) in samples from the peat site were below the test method detection limit (50, 5, 0.5, and 5 µg/L, respectively) (Exponent, 2003).

Metals with Secondary Drinking Water Standard

The following metals with secondary drinking water standards were included in the study by Humphrey and Katz (2001): aluminum (Al), iron (Fe), manganese (Mn), silver (Ag), and zinc (Zn). In addition, iron (Fe), manganese (Mn), silver (Ag), and zinc (Zn) were included in the samples from the peat site tested by Exponent (2003). The concentration of dissolved silver (Ag), and aluminum (Al) in the TDA filled trenches were generally similar to or less than background levels.

Water in direct contact with submerged TDA had elevated levels of dissolved iron (Fe), manganese (Mn), and zinc (Zn) (Table 8). Dissolved iron (Fe) concentrations in the TDA filled trenches ranged from nondetect to 86,900 µg/L compared to background concentrations of 22 to 3160 µg/L. However, with one exception in wells located 2 ft (0.6 m) downgradient of the TDA filled trench the iron (Fe) concentrations decreased to nondetect to 3660 µg/L, which is comparable to background levels.

The concentration of manganese (Mn) in the TDA filled trench was 376 to 3340 µg/L compared to background levels of 27 to 666 µg/L. For each site, the lowest manganese (Mn) concentration in the TDA filled trench was the most recent reading. There were also increased concentrations of manganese (Mn) in the downgradient wells, however, the concentration appeared to decrease with time and approach background levels.

Zinc (Zn) concentrations were also increased somewhat by TDA. The dissolved zinc (Zn) concentration in the TDA filled trench ranged from below the detection limit to 123 µg/L compared to background levels that varied from below the detection limit to 9 µg/L. However, the zinc (Zn) in the TDA filled trench decreased with time over the course of the study at two of the sites. The zinc (Zn) in downgradient wells was comparable to background levels.

Volatile Organic Compounds

Results for volatile organic compounds with the highest concentrations are summarized in Table 9. Cis-1,2-dichloroethene was found in samples from the TDA trenches on most sampling dates. Except for one sample, the concentration was below its drinking water standard of 70 µg/L. The exception had a concentration of 85.5 µg/L. The highest concentration found in the wells located down gradient of the trenches was 9.8 µg/L. The results show that TDA submerged in groundwater release low concentrations of cis-1,2-dichloroethene, however, the concentration even a short distance (2 ft; 0.6 m) down gradient of TDA was well below the compound's drinking water standard.

Table 8. Concentration of dissolved iron, manganese, and zinc at below groundwater table sites.

<i>Concentration (µg/l)</i>															
Compound	Date	Peat Site						Clay Site				Till Site			
		P1	P2	P3	P4	P5	P6	C1	C2	C3	C4	T1	T2,T3	T4	T5
		up grad	in shreds	down gradient				up grad	in shreds	down gradient		up grad	in shreds	down gradient	
Fe	6/94	514	22500	279	664	214	155	18.4	17300	21.6	<10	22	4160	2710	33
Fe	9/94	1620	66800	974	1640	1830	1180	53	56300	35	33	*	6530	*	*
Fe	11/94	*	58600	1280	1790	1700	2330	476	56400	300	318	277	71700	352	618
Fe	4/95	3160	86900	2460	2900	2190	2450	<100	#	<100	#	134	47500	<100	<100
Fe	11/96	2950	2080	*	2640	3270	3490	*	ND	*	ND	*	ND	*	28
Fe	6/97	783	49600	*	3660	3320	*	*	ND	2710	ND	*	21800	*	172
Fe	10/00	1400	80000	*	6100	*	*	*	*	*	*	*	*	*	*
Fe	11/01	1200	51000	*	2400	*	*	*	*	*	*	*	*	*	*
Mn	6/94	574	732	726	690	814	1070	120	724	322	157	49	3340	95	288
Mn	9/94	666	1340	954	786	916	845	122	1850	890	653	*	2340	*	*
Mn	11/94	*	1150	900	742	850	584	82	1400	764	44	41	2450	39	662
Mn	4/95	583	1200	1090	812	493	658	49	#	532	#	27	2500	27	773
Mn	11/96	386	514	*	518	293	484	*	502	*	29	*	973	*	40
Mn	6/97	390	619	*	228	391	*	*	376	393	13	*	780	*	56
Mn	10/00	260	570	*	300	*	*	*	*	*	*	*	*	*	*
Mn	11/01	320	530	*	240	*	*	*	*	*	*	*	*	*	*
Zn	6/94	2.8	44.4	5.1	3.3	5.4	3.8	ND	10	2.1	ND	ND	7.6	5.6	2.4
Zn	9/94	3	25	5	ND	2	2	4	123	ND	ND	*	76	*	*
Zn	11/94	*	15	8	5	2	2	7	20	4	ND	4	10	<3	23
Zn	4/95	9	15	8	ND	8	7	4	#	ND	#	4	5	5	ND
Zn	11/96	ND	65	*	8	9	ND	*	ND	*	ND	*	ND	*	ND
Zn	6/97	ND	14	*	7	6	*	*	ND	ND	ND	*	ND	*	ND
Zn	10/00	<20	90	*	<20	*	*	*	*	*	*	*	*	*	*
Zn	11/01	<20	80	*	<20	*	*	*	*	*	*	*	*	*	*

Notes: * = no sample on that date; # = compound not included in analysis on that date; ND = below detection limit (DL); see Tables 2 and 3 for detection limits. References: Humphrey and Katz (2001); Exponent (2003).

Table 9. Concentration of selected volatile organic compounds.

Concentration are in µg/L				Peat Site						Clay Site				Till Site			
Compound	Lab	Date	DL	P1	P2	P3	P4	P5	P6	C1	C2	C3	C4	T1	T2,3	T4	T5
				up grad	in shred	down gradient				up grad	in shred	down gradient		up grad	in shred	down gradient	
1,1-dichloro-ethane	C	08/94	0.5	ND	2.5	ND	ND	ND	ND	ND	1.9	6.9	ND	*	14.3	*	*
	C	11/94	0.5	*	ND	ND	ND	ND	ND	ND	7	5	ND	ND	19	ND	ND
	N	11/95	5	*	5.9	*	*	*	*	*	*	*	*	*	*	*	*
	C	08/96	0.5	ND	6.4	ND	ND	ND	*	ND	2.5	2.5	0.9	ND	12.7	3.9	ND
	N	09/96	5	*	5.8	*	*	*	*	*	<5	*	*	*	ND	*	*
	C	09/96	0.5	*	5.4	*	*	*	*	*	ND	*	*	*	0.7	*	*
	N	11/96	5	ND	<5	*	ND	ND	ND	*	<5	<5	ND	ND	12	ND	ND
	N	06/97	5	ND	<5	ND	ND	ND	*	*	ND	ND	ND	ND	*	ND	ND
	E	10/00	0.5	ND	2	*	ND	*	*	*	*	*	*	*	*	*	*
	E	11/01	0.5	ND	1.1	*	ND	*	*	*	*	*	*	*	*	*	*
4-methyl-2-pentanone	N	11/95	5	*	140	*	*	*	*	*	*	*	*	*	*	*	*
	N	09/96	5	*	40	*	*	*	*	*	21	*	*	*	ND	*	*
	N	11/96	5	ND	24	*	ND	ND	ND	*	ND	31	ND	ND	100	ND	ND
	N	06/97	5	ND	23	ND	ND	ND	*	*	ND	15	ND	ND	*	ND	ND
	E	10/00	10	ND	ND	*	ND	*	*	*	*	*	*	*	*	*	*
	E	11/01	10	ND	ND	*	ND	*	*	*	*	*	*	*	*	*	*
acetone	N	11/95	10	*	54	*	*	*	*	*	*	*	*	*	*	*	*
	N	09/96	10	*	21	*	*	*	*	*	10	*	*	*	ND	*	*
	N	11/96	10	ND	ND	*	ND	ND	ND	*	ND	ND	ND	ND	40	ND	10
	N	06/97	10	ND	13	ND	ND	ND	*	*	ND	ND	ND	ND	*	ND	ND
	E	10/00	10	ND	ND	*	ND	*	*	*	*	*	*	*	*	*	*
	E	11/01	10	ND	27	*	11	*	*	*	*	*	*	*	*	*	*
benzene	C	08/94	0.5	ND	0.7	ND	ND	ND	ND	ND	1.4	ND	ND	*	1.8	*	*
	C	11/94	0.5	*	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
	N	11/95	5	*	ND	*	*	*	*	*	*	*	*	*	*	*	*
	C	08/96	0.5	ND	1.8	ND	ND	ND	*	ND	2	1	ND	ND	1.5	ND	ND
	N	09/96	5	*	<5	*	*	*	*	*	<5	*	*	*	ND	*	*
	C	09/96	0.5	*	21	*	*	*	*	*	9.5	*	*	*	ND	*	*
	N	11/96	5	ND	<5	*	ND	ND	ND	*	ND	<5	ND	ND	ND	ND	ND
	N	06/97	5	ND	ND	ND	ND	ND	*	*	ND	ND	ND	ND	*	ND	ND
	E	10/00	0.5	ND	1.1	*	ND	*	*	*	*	*	*	*	*	*	*
	E	11/01	0.5	ND	1.6	*	ND	*	*	*	*	*	*	*	*	*	*
chloroethane	C	08/94	1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	*	1.1	*	*
	C	11/94	1	*	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
	N	11/95	10	*	ND	*	*	*	*	*	*	*	*	*	*	*	*
	C	08/96	1	ND	ND	ND	ND	ND	*	ND	ND	ND	ND	ND	ND	ND	ND
	N	09/96	10	*	<10	*	*	*	*	*	ND	*	*	*	ND	*	*
	C	09/96	1	*	6.7	*	*	*	*	*	ND	*	*	*	ND	*	*
	N	11/96	10	ND	<5	*	ND	ND	ND	*	ND	ND	ND	ND	ND	ND	ND
	N	06/97	10	ND	ND	ND	ND	ND	*	*	ND	ND	ND	ND	*	ND	ND
	E	10/00	1	ND	2	*	ND	*	*	*	*	*	*	*	*	*	*
	E	11/01	1	ND	ND	*	ND	*	*	*	*	*	*	*	*	*	*

Table 9 (continued). Concentration of selected volatile organic compounds.

Concentration are in µg/L				Peat Site						Clay Site				Till Site			
Compound	Lab	Date	DL	P1	P2	P3	P4	P5	P6	C1	C2	C3	C4	T1	T2,3	T4	T5
				up grad	in shred	down gradient				up grad	in shred	down gradient		up grad	in shred	down gradient	
cis-1,2-di-chloroethene	C	08/94	0.5	ND	16.1	ND	ND	ND	ND	ND	9.2	ND	ND	*	33.4	*	*
	C	11/94	0.5	*	6	ND	ND	ND	ND	ND	34.5	8.5	ND	ND	85.5	ND	ND
	N	11/95	5	*	25	*	*	*	*	*	*	*	*	*	*	*	*
	C	08/96	0.5	ND	36.9		ND	4.2		ND	9.4	7	0.8	ND	44.7	9.8	ND
	N	09/96	5	*	29	*	*	*	*	*	6.4	*	*	*	ND	*	*
	C	09/96	0.5	*	32.2	*	*	*	*	*	7.9	*	*	*	1	*	*
	N	11/96	5	ND	26	*	ND	ND	ND	*	6	7.9	ND	ND	43	ND	ND
	N	06/97	5	ND	27	<5	ND	ND	*	*	<5	6	ND	ND	*	<5	ND
	E	10/00	0.5	ND	16	*	1.1	*	*	*	*	*	*	*	*	*	*
	E	11/01	0.5	ND	8.1	*	ND	*	*	*	*	*	*	*	*	*	*

Notes: * = no sample on that date; ND = below detection limit; DL = method detection limit; C = sample tested by University of Connecticut; N = sample tested by Northeast Laboratory; E = sample tested by Exponent (2003)

Benzene appears to be released from TDA at trace levels. The measured concentrations were less than the drinking water standard (5 µg/L) except for two samples tested by the University of Connecticut. Duplicate samples tested by Northeast Laboratory found concentrations less than 5 µg/L. The concentration of benzene in wells 2 ft (0.6 m) down gradient of the trench were generally below detection limits except for two samples that had concentrations of 1 and <5 µg/L.

TDA releases low levels of 1,1-dichloroethane, 4-methyl-2-pentanone (MIBK), and acetone. The concentration 1,1-dichloroethane in samples taken from the TDA trenches ranged from nondetect to 19 µg/L. However, the concentrations in the wells down gradient from the trenches were below the detection limit in the most recent round of sampling. Methyl-2-pentanone (MIBK) was found in samples from the TDA trenches at concentrations ranging from nondetect to 140 µg/L, however the highest concentration in a well located 2 ft (0.6 m) down gradient from the trench was 31 µg/L. The concentration of acetone in the TDA trenches ranged from nondetect to 54 µg/L, however, this compound is naturally produced by human metabolism and is not a major health concern at low concentrations. TDA appears to release trace levels of 1,1,1-trichloroethane, 1,1-dichloroethene, xylenes, toluene, trichloroethene, 2-butanone (MEK), and chloroethane. The concentrations are generally less than 10 µg/L. For compounds that have a drinking water standard, the levels were well below the standard (USEPA, 2006).

Semivolatile Organic Compounds

TDA submerged in groundwater release low levels of aniline (Table 10). When detected, the concentrations ranged from 20 to 200 µg/L. Phenol was found in the samples taken from the TDA trenches at all three sites in November, 1994, and April, 1995. The concentrations ranged from 16 to 55 µg/L. The levels were nondetect in the first round of samples in August, 1994, and in all samples taken on or after November, 1995. More importantly, no detectable levels of either compound were found in the down gradient wells.

m&p cresol was present above the detection limit in slightly more than half of the samples taken from the TDA trenches. When detected, concentrations ranged from 13 to 86 µg/L. Except for two samples at the clay site, neither compound was found in the down gradient wells. Thus, m&p cresol is released at low concentrations, however, it has a negligible tendency to migrate down gradient.

It is likely that TDA release trace levels of benzoic acid and N-nitrosodiphenylamine. Benzoic acid was found in about one-quarter of the samples taken from the TDA trenches at concentrations ranging from <10 to 100 µg/L. N-nitrosodiphenylamine was found in about one-third of the samples taken from the TDA trenches at concentrations ranging from <10 to 11.2 µg/L. Except for one sample each, these compounds were not found in the down gradient wells.

The following compounds were found above the detection limit in a few samples: benzothiazole, 2(3H)-benzothiazolone, 3-methylbenzenamine, and di-n-butyl-phthate. It is possible that TDA sporadically release low levels of these compounds. Several compounds were reported at levels above the detection limit in one well on a single date. This includes: cyclohexanol, 2,6-bis-(1,1-dimethylethyl)-2,5-cyclohexadiene-1,4-dione, 1H-isoindeole-1,3(2H)-dione, 4-(2-benzothiazolythio)-morpholine, N-(1,1-dimethylethyl)-formamide, butanoic acid, and isothiocyanato cyclohexane. It is possible that TDA sporadically release low levels of these compounds. However, it is more likely that these are spurious data points and that TDA does not release detectable levels of these compounds.

Nine semi-volatile compounds were tentatively identified on a single date in the tests performed by the University of Connecticut. The compounds are: cyclohexanol, 2,6-bis-(1,1-dimethylethyl)-2, 1H-isoindeole-1,3(2H)-dione, 4-(2-benzothiazolythio)-morpholine, N-(1,1-dimethylethyl)-formamide, butanoic acid, diethyltoluamide (DEET), 3-methylbenzenamine, and isothiocyanato cyclohexane. In addition, benzothiazole was tentatively identified on three sampling dates and 2(3H)-benzothiazolone was tentatively identified on two sampling dates. There were also unidentified compounds on two dates. The estimated concentrations of these compounds are given in Table 10. It is possible that TDA releases low concentrations of these compounds (Humphrey and Katz, 2001).

Table 10. Concentration of selected semivolatile organic compounds.

Concentration in µg/L				Peat Site						Clay Site				Till Site			
Compound	Lab	date	DL	P1	P2	P3	P4	P5	P6	C1	C2	C3	C4	T1	T2,3	T4	T5
				up gr.	in shred	down gradient				up gr.	in shred	down gradient		up gr.	in shred	down gradient	
aniline	C	8/94	10	ND	58	ND	ND	ND	ND	ND	*	ND	*	*	31	*	*
aniline	C	11/94	10	*	20	ND	ND	ND	ND	ND	91	ND	ND	*	64	ND	ND
aniline	C	4/95	--	#	#	#	#	#	#	#	#	#	#	#	(40)	#	*
aniline	N	11/95	10	*	81	*	*	*	*	*	*	*	*	*	*	*	*
aniline	C	11/95	--	*	(30)	*	*	#	*	*	*	*	*	*	*	*	*
aniline	C	7/96	--	#	#	*	#	#	#	#	#	#	#	#	(100)	#	#
aniline	N	10/96	10	*	ND	*	*	*	*	*	ND	*	*	*	ND	*	*
aniline	C	10/96	--	*	(200)	*	*	*	*	*	(65)	*	*	*	#	*	*
aniline	N	11/96	10	ND	ND	*	ND	ND	ND	*	ND	ND	ND	*	ND	ND	ND
aniline	N	6/97	10	ND	ND	*	ND	ND	ND	*	ND	ND	ND	ND	ND	ND	ND
aniline	E	10/00	5	ND	100	*	ND	*	*	*	*	*	*	*	*	*	*
aniline	E	11/00	5	ND	ND	*	ND	*	*	*	*	*	*	*	*	*	*
phenol	C	8/94	20	ND	ND	ND	ND	ND	ND	ND	*	ND	*	*	ND	*	*
phenol	C	11/94	20	*	55	ND	ND	ND	ND	ND	16	ND	ND	*	26	ND	ND
phenol	C	4/95	20	ND	27	ND	ND	ND	ND	ND	22	ND	ND	ND	51	ND	*
phenol	N	11/95	10	*	ND	*	*	*	*	*	*	*	*	*	*	*	*
phenol	C	11/95	20	*	ND	*	*	ND	*	*	*	*	*	*	*	*	*
phenol	C	7/96	20	ND	ND	*	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
phenol	N	10/96	10	*	ND	*	*	*	*	*	ND	*	*	*	ND	*	*
phenol	C	10/96	20	*	ND	*	*	*	*	*	ND	*	*	*	ND	*	*
phenol	N	11/96	10	ND	ND	*	ND	ND	ND	*	ND	ND	ND	*	ND	ND	ND
phenol	N	6/97	10	ND	ND	*	ND	ND	ND	*	ND	ND	ND	ND	ND	ND	ND
phenol	E	10/00	5	ND	ND	*	ND	*	*	*	*	*	*	*	*	*	*
phenol	E	11/00	5	ND	ND	*	ND	*	*	*	*	*	*	*	*	*	*
m&p cresol	C	8/94	20	ND	ND	ND	ND	ND	ND	ND	*	ND	*	*	ND	*	*
m&p cresol	C	11/94	20	*	ND	ND	ND	ND	ND	ND	ND	ND	ND	*	ND	ND	ND
m&p cresol	C	4/95	20	ND	32	ND	ND	ND	ND	ND	42	ND	ND	ND	86	ND	*
m&p cresol	N	11/95	10	*	13	*	*	*	*	*	*	*	*	*	*	*	*
m&p cresol	C	11/95	20	*	20	*	*	ND	*	*	*	*	*	*	*	*	*
m&p cresol	C	7/96	20	ND	29	*	ND	ND	ND	ND	39	ND	ND	ND	82	ND	ND
m&p cresol	N	10/96	10	*	14	*	*	*	*	*	31	*	*	*	ND	*	*
m&p cresol	C	10/96	20	*	42	*	*	*	*	*	24	*	*	*	ND	*	*
m&p cresol	N	11/96	10	ND	ND	*	ND	ND	ND	*	ND	18	ND	*	57	ND	ND
m&p cresol	N	6/97	10	ND	ND	*	ND	ND	ND	*	ND	<10	ND	ND	32	ND	ND
m&p cresol	E	10/00	5	ND	ND	*	ND	*	*	*	*	*	*	*	*	*	*
m&p cresol	E	11/00	5	ND	ND	*	ND	*	*	*	*	*	*	*	*	*	*
benzothiazole	C	4/95	--	#	#	#	#	#	#	#	(50)	#	#	#	#	#	*
benzothiazole	C	11/95	--	*	(100)	*	*	#	*	*	*	*	*	*	*	*	*
benzothiazole	C	10/96	--	*	(300)	*	*	*	*	*	(300)	*	*	*	#	*	*

Table 10 (continued). Concentration of selected semivolatile organic compounds.

Concentration in µg/L				Peat Site						Clay Site				Till Site			
Compound	Lab	date	DL	P1	P2	P3	P4	P5	P6	C1	C2	C3	C4	T1	T2,3	T4	T5
				up gr.	in shred	down gradient				up gr.	in shred	down gradient		up gr.	in shred	down gradient	
2(3H)-benzo- -thiazolone	C	4/95	--	#	(200)	#	#	#	#	#	(100)	(100)	#	#	(100)	#	*
	C	10/96	--	*	(300)	*	*	*	*	*	(300)	*	*	*	#	*	*
Unknown	C	4/95	--	#	(700)	#	#	#	#	#	(40)	(480)	(239)	#	(600)	(150)	#
Unknown	C	4/95	--	#	(200)	#	#	#	#	#	#	#	#	#	#	#	#

Notes: * = no sample on that date; # = compound not included in analysis on that date; ND = below detection limit; ()= tentatively identified compound; DL = method detection limit; C = sample tested by University of Connecticut; N = sample tested by Northeast Laboratory; E = sample tested by Exponent; References: Humphrey and Katz (2001); Exponent (2003)

Aquatic Toxicity

Samples from the peat site were used to perform two USEPA (USEPA, 1991) freshwater short-term toxicity tests. One test was three-brood survival and reproduction of the crustacean *Ceriodaphnia dubia*. In this test, females are exposed to samples for the period of time it takes for control treatment females to produce three broods, typically about 7 days. Then survival and reproduction of the test and control groups are compared. The other test was seven-day survival and growth of larval fathead minnows (*Pimephales promelas*). At the completion of the test, survival and growth for minnows exposed to field samples are compared to control samples. Samples were taken on two dates: 10/9/2000 and 11/11/2001. On each date samples were taken from the control well (P1), TDA filled trench (P2), and 2 ft (0.6 m) downgradient (P4).

On both sample dates there was no effect on survival of fathead minnows in samples from P1, P2, and P4 in undiluted leachate. There was a slight effect on growth of fathead minnows for leachate from each well as shown in Table 11. Leachate from the TDA filled trench had to be diluted to between 77% and 91% of “full strength” to reach growth comparable to the laboratory control. However, slightly greater dilution was required for naturally occurring groundwater (P1). Thus, the survival and growth of fathead minnows in water from the TDA filled trench and natural groundwater at the site were comparable.

Table 11. Chronic toxicity of leachate waters to fathead minnows – effect on growth (Exponent, 2003).

Sample	10/09/2000	11/11/2001
P1 (groundwater)	69% leachate	70% leachate
P2 (TDA filled trench)	77% leachate	91% leachate
P4 (2 ft downgradient)	89% leachate	78% leachate

The survival and reproduction effects of leachate on *Ceriodaphnia dubia* are summarized in Table 12. There was a slight effect on survival (P2 and P4) as compared to groundwater (P1) for 10/09/00. However, survival occurred in undiluted samples from all wells for 11/11/01. To reach the target reproduction, greater dilution was required for P2 and P4 as compared to groundwater (P1). A maximum of 4-fold dilution would be required of water from P2 to reach conditions of the groundwater at the site (P1). After flowing through 2 ft (0.6 m) of soil to P4, the amount of dilution needed to reach that of groundwater had reduced to 2.6-fold. The reason for the toxic effects was thought to be precipitation of iron on the breathing apparatus of the *Ceriodaphnia dubia* (Exponent, 2003; Sheehan, et al., 2006). Thus, while there were some toxic effects of TDA placed below the groundwater table on *Ceriodaphnia dubia*, a small amount of dilution as the groundwater flowed down gradient or when it entered a surface body of water would remove the toxic effects.

Table 12. Dilution of leachate needed to reach survival and reproduction targets for *Ceriodaphnia dubia* (Exponent, 2003).

Sample	Survival (EC ₅₀)		Reproduction (IC ₅₀)	
	10/09/00	11/11/01	10/09/00	11/11/01
P1 (groundwater)	>100%	>100%	66%	70%
P2 (TDA filled trench)	66%	>100%	16%	28%
P4 (2 ft downgradient)	77%	>100%	25%	54%

Summary of Water Quality Effects of TDA Placed Below the Groundwater Table

The preponderance of evidence shows that TDA does not cause metals with primary drinking water standards to be exceeded. Moreover, the data shows that TDA is unlikely to increase levels of metals with primary drinking water standards above naturally occurring background levels.

For chemicals with secondary drinking water standards, it is likely that TDA below the groundwater table would increase the concentrations of iron, manganese, and zinc. For water that is collected directly from TDA fill below the groundwater table, it is likely that the concentrations of manganese and iron will exceed their secondary drinking water standards. However, the concentration decreases to near background levels by flowing only a short distance through soil. For other chemicals with secondary drinking water standards, there is no evidence that TDA affects naturally occurring background levels.

Trace levels of a few volatile and semivolatile organics were found from water taken directly from TDA filled trenches. However, concentrations were below method detection limits for virtually all samples taken from wells 2 ft (0.6 m) and 10 ft (10 ft) downgradient.

Aquatic toxicity tests were performed on samples taken on two dates. The results showed that water collected directly from TDA filled trenches had no effect on survival, and growth of fathead minnows. While there were some toxic effects of TDA placed

below the groundwater table on *Ceriodaphnia dubia*, a small amount of dilution as the groundwater flowed down gradient or when it entered a surface body of water would remove the toxic effects.

In summary, TDA placed below the water table would be expected to have a negligible off-site effect on water quality.

WATER QUALITY EFFECTS FOR SPECIFIC APPLICATIONS

The water quality effects of TDA used in specific applications as well as RMA pavement will be discussed in the following sections. Much of this discussion will make use of the data for TDA placed above and below the water table presented in the previous sections. Where appropriate, additional application specific data will be introduced.

TDA as Fill for Roadway, Wall, Building, and Railroad Construction

In this category of applications, TDA is used as lightweight fill for embankments constructed on weak ground, lightweight backfill for walls and bridge abutments, insulation and drainage layers beneath roads, free-draining and insulating backfill for residential foundations, and vibration damping layers beneath rail lines. The TDA is generally 3 to 12-in. (75 mm to 300 mm) maximum size. In most of applications in this category, the TDA is placed above the water table, but there are some instances where the TDA will be placed below the water table. The data presented previously on TDA above and below the water table is directly applicable to these circumstances.

For both above and below water table applications, the preponderance of evidence shows that TDA will not cause primary drinking water standards to be exceeded. Moreover, TDA is unlikely to increase levels of metals with primary drinking water standards above naturally occurring background levels.

For metals with a secondary drinking standard, it is likely that water in direct contact with TDA, either in above or below the groundwater table applications, will have elevated concentrations of dissolved iron and manganese. Moreover, the secondary drinking water standards for iron and manganese will likely be exceeded. The concentration of iron and manganese is higher for below groundwater table applications. In addition, for water in direct contact with TDA placed below the groundwater table, it is likely that the concentration of zinc will be increased, but to levels below the applicable secondary drinking water standard. For five out of six field sites with monitoring wells adjacent to the TDA fill, it was found that flow through soil for a distance of 2 to 10 ft (0.6 to 3 m) was generally adequate to attenuate the concentration of iron, manganese and zinc to near naturally occurring background levels. Thus, the downgradient mobility of iron, manganese, and zinc is limited. For other chemicals with secondary drinking water standards, there is no evidence that TDA affects naturally occurring background levels. Secondary standards are “non-enforceable guidelines regulating contaminants that may cause cosmetic effects (such as skin or tooth discoloration) or aesthetic effects (such as taste, odor, or color) in drinking water” (USEPA, 2006), so these are not health based concerns.

Volatile and semivolatile organic compounds were monitored for two sites where TDA was placed above the water table and for three sites where TDA was placed below the water table. For water in direct contact with TDA, a few compounds are found above the detection limit. The concentrations tended to be slightly higher for below groundwater table sites. For those compounds with drinking water standards, the concentrations were below the corresponding standard with the exception of one compound on one sample date at a below groundwater table site. Measurements taken at the three sites where TDA was placed below the groundwater table show that flow through as little as 2 ft (0.6 m) of soil generally reduces concentrations of monitored compounds to below the test method detection limits.

Freshwater toxicity tests showed that undiluted leachate from TDA fills placed above the water table had no effect on survival, growth, and reproduction of two standard test species (fathead minnows and a small crustacean, *Ceriodaphnia dubia*). For undiluted leachate from TDA placed below the groundwater table, there was no effect on survival and growth of fathead minnows. However, there was some toxic effects of undiluted leachate on survival and reproduction of *Ceriodaphnia dubia*. About a 4-fold dilution is needed to reach the no effect level. For most projects this would occur due to mixing with groundwater downgradient from the project or with surface water.

Based on these results, TDA used as fill in above groundwater table applications would be expected to have negligible human health based effects on drinking water quality or aquatic toxicity effects on fresh water species. Except for immediately adjacent to TDA fills, there would be negligible impact on secondary (aesthetic) drinking water standards. In the limited cases where it is desirable to use TDA as fill below the water table, the results show that there would be negligible human health based effects on drinking water quality. Immediately downgradient of the TDA fill, there would be some secondary effects on drinking water quality and there could be some aquatic toxicity effects. Thus, for TDA fill that is submerged below the groundwater table for most of the year, a reasonable separation should be maintained between the TDA fill and shallow drinking water wells or surface bodies of water with sensitive aquatic species.

TDA Used in Landfill Applications

Use of TDA in landfill applications can be divided into three categories:

- As a component that is within the lined landfill itself such as: leachate collection layer, leachate collection system protection layer, gas collection layers and trenches, and leachate recirculation trenches.
- As a component that is outside of the lined landfill itself such as: a drainage layer in the landfill cap or French drain around the perimeter of a landfill to control groundwater.
- Special cases where the TDA is within a lined landfill cell, but no waste has yet been placed in the cell.

In the first case, any chemicals that are released by the TDA will become commingled with leachate from the waste. The leachate will then be collected and removed by the leachate collection and removal system, and subsequently treated and disposed of in accordance with rules and regulations applicable to landfill leachate. In one field study, TDA was found to increase the levels of iron and zinc, and to a lesser extent copper, chromium, and barium (Aydilek, et al., 2006). This study concluded that TDA can be used as a component in landfill leachate collection systems.

In the second case, the TDA will be subjected to infiltration from rainwater and snowmelt. Moreover, the water that has passed through the TDA will not be collected by the landfill leachate collection and removal system. In this case, the results for TDA fills placed above the water table, as discussed in the previous section are directly applicable and will not be repeated here. It should be noted that wells for drinking water will never be located in close proximity to a landfill and the landfill will have a surface water control plan.

In the third case, the TDA would be in place as a component of the leachate collection and removal system (LCRS) or the protection layer over the LCRS, but waste had not yet been placed in the cell. During this period of time, rainwater that falls into the cell would be collected and removed. The landfill operator may desire to dispose of the water rather than subjecting it to the treatment protocols required for leachate. In this case, the water quality effects would be expected to be similar to TDA placed above the groundwater table discussed in the previous section.

In general, TDA would have minimal water quality effects when used in landfill applications.

TDA Used as Drain-Field Aggregate for Septic Systems

TDA is used as the aggregate in drain fields for on-site waste water treatment systems, commonly referred to as septic systems. The data for TDA placed above the water table are useful for assessing this application. These results would suggest that TDA used as drain field aggregate would leach negligible levels of compounds with primary drinking water standards, some iron and manganese, and negligible levels of organics. These results were confirmed by two field trials where use of TDA was compared to conventional gravel aggregate (Miller and Chadik, 1993; Sengupta and Miller, 2004).

Rubber Modified Asphalt Pavements

Two laboratory studies have been conducted to evaluate the water quality effects of rubber modified asphalt pavements (Vashisth, et al., 1998; Wright, et al., 1999; Azizian, et al., 2003). The study by Wright, et al. (1999) included leaching tests on compacted rubber modified asphalt samples and flat plate leaching tests that simulated a rainfall event. Most of the results presented by Azizian, et al. (2003) were batch leaching tests in which the rubber modified asphalt was reduced to particles smaller than

¼-in. in size prior to leachate extraction. It is unlikely that the results from these studies replicate field conditions.

The study by Wright, et al. (1999) and Vashisth, et al. (1998) found that crumb rubber modified asphalt samples leached higher levels of some metals and organic compounds compared to samples of conventional hot mix asphalt pavement. Azizian, et al. (2003) found similar results. The latter study went on to show that the organic contaminant with the highest concentration, benzothiazole, was volatile and biodegradable. Moreover, the metals and organics were sorbed onto soil such that the leachate was rendered nontoxic to freshwater aquatic species.

SUMMARY

Tire derived leachate (TDA) has a limited effect on drinking water quality and fresh water aquatic toxicity for a range of applications including lightweight backfill for walls and bridge abutments, insulation and drainage layers beneath roads, free-draining and insulating backfill for residential foundations, vibration damping layers beneath rail lines, landfill leachate collection systems, drainage layers in landfill caps, landfill gas collection systems, and drainage aggregate for drain fields for on-site waste water treatment systems. TDA is unlikely to increase the concentration of substances with primary drinking water standards above those naturally occurring in the groundwater. It is likely that TDA will increase the concentration of iron and manganese, but the data indicates that these elements have limited ability to migrate away from the TDA installation. TDA placed above the water table has negligible toxic effects for fresh water aquatic organisms. Undiluted leachate from TDA placed below the water table would have some toxic effects, but with only a small amount of dilution the effects are reduced to negligible levels.

The body of work examining the drinking water quality and aquatic toxicity effects of rubber modified asphalt pavements is limited. However, the available work suggests that rubber modified asphalt pavements would have negligible off-site effects on water quality and toxicity.

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An experimental test embankment was built at a landfill near Madison, Wisconsin using shredded waste tires. Prior to construction of the embankment, toxicity and leaching tests were performed to determine possible environmental risks, and the tests showed that the tire shreds were not a hazardous material. In construction of the embankment, two leachate collection lysimeters were placed beneath areas containing tire shreds. Each lysimeter was made of a 10-mil-thick PVC liner, and was covered with a nonwoven geotextile. A PVC pipe directed the leachate into a cylindrical PVC container. Water quality analysis was conducted on leachate collected by the lysimeters. The pH was found to be stable at around 7.5. A high manganese content was found, but groundwater and surface water data gathered from the Dane County Public Works Department showed that there was a history of high manganese concentrations in the areas around the test embankment. From the study of the experimental test embankment, it was determined that the possibility of tire shreds affecting groundwater quality is highly unlikely.

Brophy, M. & Graney, J. (2004). Groundwater Effects from Highway Tire Shred Use. *Environmental Forensics*, Vol. 5, No. 2, pp. 79-84.

Tire shreds were used in construction of the subgrade for an interstate exit ramp in New York State. Three groundwater monitoring wells and two tire fill sampling ports were installed. One of the monitoring wells was placed upgradient from the tire shreds, while the other two wells were positioned downgradient. One of the sampling ports was placed directly below the tire shred fill area, and the other port was placed adjacent to the tire shred fill area. Both sampling ports were situated between the exit ramp and a man-made detention pond. The average pH for the downgradient wells and port beneath the fill area were lower than the average pH in the upgradient well and adjacent port. The same trend held true for dissolved oxygen content. The sampling port beneath the tire fill area was the only site at which organic compounds were detected. The concentrations were low and any organic contaminants were believed to have come from highway runoff or cleaning substances, rather than from the tire shreds themselves. In testing for metals, iron, manganese, barium, cadmium, and zinc were detected at concentrations above accepted levels. Zinc was found in the upgradient well, suggesting that the metal may have been from natural sources. Iron and manganese were present in the upgradient well,

but significantly higher concentrations were found downgradient. This was also the case for barium, but the levels of barium present were far below the accepted level. Cadmium was not detected upgradient but was found in the downgradient well. These findings suggest that some metal may have been leached from the tire shreds.

Chyi, L.L. (2000). Shredded Tire Leachate from Monofill Sites. *Department of Geology, The University of Akron*, 131 p.

In this study, leachate was collected from two Stark County, Ohio monofills. The monofills contain only tire shreds and are located in abandoned coal strip mines. The monofills have leachate collection systems so the results are applicable to above groundwater table conditions. The size and other characteristics of the tire shreds were not provided. Some of the samples were taken directly from the leachate collected at the bottom of the landfill. The reported results are for filtered samples. The samples were analyzed for twelve commonly occurring cations and anions, as well as, an additional 19 trace elements. Results for total dissolved solids, pH, conductivity, and alkalinity were also reported. Manganese, sulfate, and total dissolved solids were detected at concentrations higher than World Health Organization (WHO) limits at both sites. Calcium, magnesium, sodium, and potassium were also components of the leachate from both sites, and the pH at each site was lower than comparison values found in a laboratory study on crumb rubber. Selected results are compared to area groundwater. Iron concentrations were as much as 100 times higher than comparison groundwater data. Manganese concentrations were also higher than naturally occurring in groundwater. For trace elements, the average concentration of arsenic, cadmium, nickel, lead, antimony, and selenium at one or both sites exceeded WHO and EPA drinking water standards. However, the averages were computed using only values that were above the detection limit. This caused the reported average to be too high for any parameter that had one or more results below the detection limit. Moreover, it was not possible to determine the influence of the site geology and groundwater chemistry on these results. It was concluded that a leak in the monofill liner could elevated levels of some cations and anions in the groundwater.

Eldin, N.N. and Senouci, A.B. (1992). Use of Scrap Tires in Road Construction. *Journal of Construction Engineering and Management*, Vol. 118, No. 3, pp. 561-576.

A portion of this field study focused on the impacts of tire shreds on ground water quality. A 120 ft test road was divided up into six 20 ft sections to test different combinations of tire shred size, different mixtures of tire shreds and soil, soil cover thickness, and tire shred placement method. Pan lysimeters were placed beneath sections 2 and 5 to collect rainwater that would leach through the tire shred layer. Road section 2

had 100% 4 in. tire shreds, placed in 1 ft lifts with a 1 ft soil cover thickness. Road section 5 was constructed in the same manner, but with 2 in. rather than 4 in. shreds. The soil cover had a pH of 6.9. Samples were collected monthly over a period of 6 months and tested for ten inorganic constituents. In addition, five water quality index tests were performed. It was found that hazardous limits of Ba and Pb were not exceeded. Preventive action limits (PAL) were exceeded by barium, calcium, magnesium, manganese, sodium, alkalinity, hardness, BOD, and COD in all cases. Concentrations of lead were below PAL in section 2, but were higher in section 5. The study suggests that this variation could be due to the smaller shred size and thus increased surface area in section 5. Fe concentrations fluctuated above and below PAL. SO₄ exceeded PAL but decreased with time. It is noted in the study that PAL would normally be enforced for leachate obtained some distance from the roadway as opposed to directly underneath the road. Therefore, the leaching elements would tend to dilute with distance from the source, thus decreasing the measured concentrations. It is also noted that PAL are only guidelines which suggest that the situation may need evaluation to determine if preventative action should be initiated. The study uses the environmental data to conclude that tire shreds, when used in a similar manner as in the test site, should not have a harmful effect on the ground water quality, but the study also recognizes that laboratory tests and more field tests should be done.

Hoppe, E.J. (1998). Field Study of Shredded-Tire Embankment. *Transportation Research Record 1619*, Transportation Research Board, Washington, D.C., 47-54.

In this study, two highway embankments were constructed in York County, Virginia in 1993. About 1.7 million tires were used in the project. The tire shred portions of the embankments consisted of 50% tires and 50% silty sand and were 80 m wide by 160 m long and 6 m high, capped with 1.5 m of soil. The shreds used were a maximum of 25 cm, did not have both sidewalls attached, and contained no free steel strands. In accordance with the Virginia Solid Waste Management Regulations, a well was placed beside one of the embankments, and another well was placed upstream from the embankments so as to serve as a control. By comparing the groundwater collected from both of the wells, it was determined that calcium, magnesium, sodium, and chloride were not leaching from the tire shreds. It was also observed that the concentrations of iron, lead, and zinc in the embankment well were not statistically higher than in the control well. Hardness, pH, total organic carbon, and total organic halides were not affected by the presence of tire shreds. There were also no indications of exothermic reactions in the embankments. It was concluded that the use of tire shreds in highway embankments has no harmful effects on the environment.

Humphrey, D.N. (1999). *Water Quality Results for Whitter Farm Road Tire Shred Field Trial*. Dept. of Civil and Environmental Engineering, University of Maine, Orono, Maine, 6 p.

A field trial was conducted to analyze insulation and drainage properties of a tire shred layer placed below a paved road, as well as to collect data on the effects of tire shreds on water quality. The Whitter Farm Road tire shred field trial was constructed near the University of Maine in Orono, Maine using tire shreds with a maximum size of 3 in. (76 mm). The shreds had exposed steel and glass wire. A 6.5-ft (2.0-m) deep edge drain, backfilled with tire shreds or mixtures of tire shreds and soil, was located beneath one shoulder. Water discharging from the edge drain was sampled on one date and analyzed for metals and organic compounds. Cadmium, chromium, copper, lead, and barium (metals with a primary drinking water standard) had dissolved and total concentrations below their maximum limits. Aluminum, chloride, iron, sulfate, and zinc (metals with an aesthetic based secondary drinking water standard) were also below their dissolved concentration limits. Iron had a total concentration above the standard due to iron oxide particles. Manganese, which also has a secondary drinking water standard, had a dissolved concentration above its standard level. This was probably due to the fact that the steel belts in the tires contain 2-3% manganese by weight. Both volatile and semivolatile organic compound levels were lower than test method detection limits. The results of this field trial demonstrate that tire shreds in place above the water table have an insignificant effect on the groundwater quality.

Humphrey, D.N. and Katz, L.E. (2000). Water-quality effects of tire shreds placed above the water table: Five-year field study. *Transportation Research Record 1714*, 18-24.

This field trial, which was built on Route 231 in North Yarmouth, Maine, was studied from January 1994 to June 1999 to determine long-term water quality effects of tire shreds placed above the water table. The site consisted of five 100-ft (33-m) long sections, four of which were constructed using a 2-ft (0.61-m) -thick layer of tire shreds topped with 2.5 to 4.5 ft (0.76 to 1.37 m) of granular soil and 0.42 ft (0.127 m) of pavement. The fifth section was established as a control section with conventional soil fill, since many of the inorganic substances to be tested for are commonly found to some extent in groundwater. About 100,000 tires shredded to a maximum size of 3 in. (75 mm) were used. Two high-density polyethylene-lined seepage collection basins were installed under the sections containing tire shreds, and another basin was installed under the control area. Organic substances, inorganic substances with primary and secondary drinking water standards, pH, and other water quality index parameters were tested for. The pH levels were not significantly affected by the tire shreds, and the levels of organic substances were found to be negligible. The study did not find any evidence that the concentrations of substances with a primary drinking water standard, including barium, cadmium, chromium, copper, lead, and selenium, were increased by the presence of tire

shreds. Aluminum, chloride, sulfate, and zinc, which have secondary drinking water standards, were also not affected by the tire shreds. It was determined that manganese and iron levels have the potential to exceed secondary drinking water standards, but these substances are only of aesthetic concern. The study concludes that tire shreds can be used as lightweight fill, backfill, and insulation on a variety of highway projects.

Humphrey, D.N., and Katz, L.E. (1995). Water Quality Testing for Dingley Road Tire Chip Test Project. *A Report for the Town of Richmond, Maine, by Department of Civil and Environmental Engineering, University of Maine, Orono, Maine*, 35 p.

This report details a 950 ft-long, five-section test site on Dingley Road in Richmond, Maine. Tire shred layers of 6 in. and 12 in. were used, and granular soil layers of 12 in., 18 in., and 24 in. were placed over the tire shreds in different sections. About 20,000 tires were shredded to a maximum shred size of 2 in. (51 mm) for use on this project. Six PVC wells, including one positioned at a control site, were installed at the test site to collect groundwater. Water samples were taken on three occasions from December 1993 to January 1995. None of the tested substances with primary drinking water standards were found in concentrations that exceeded those standards. Six substances with secondary drinking water standards were tested for. Manganese was found to exceed the secondary standard in the areas with tire shreds, as well as in the control, so it is suggested that manganese was naturally present in the groundwater. Levels of calcium, magnesium, and sodium typical of Maine drinking water were detected. The study concludes that significant levels of inorganic contaminants had not leached from the tire shreds in the 28 months following construction of the road, but more studies would need to be done to determine long-term effects of the tire shreds.

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Barris, D.C. (1987). Report of Ground & Surface Water Analyses. *Environmental Consulting Laboratory*.

This unpublished report provides the results of water quality analyses from Hamden Tire Salvage in Hamden, Connecticut. Results for a single sampling date, May 15, 1987, are reported. Although not stated in the report, this site contains whole tires that had been placed in a pond formed in an abandoned quarry. Water samples were taken directly from the pond containing the tires and three sampling wells. The proximity of the sampling wells to the tire-filled pond was not given. The samples were tested for five metals, six pesticide/herbicides, 36 volatile organic compounds (EPA Method 8010), and 15 aromatic volatile organic compounds (EPA Method 8020). In addition, ten water

quality index tests were performed. The sample from the pond had a higher pH, conductivity, and total dissolved solids than the samples from the wells. Pesticides/herbicides were below the method detection limit for all samples. Three volatile organic compounds (trans-1,2-dichloroethylene, 1,2-dichloropropane, and trichloroethylene) were above the detection limit in the sample from one of the wells. Toluene and mixed xylenes were above the detection limit in the sample from the pond. Results for other organic compounds were below the detection limit.

Humphrey, D.N. and Katz, L.E. Field Study of the Water Quality Effects of Tire Shreds Placed Below the Water Table. *Proceedings of the International Conference on Beneficial Use of Recycled Materials in Transportation Applications, November 13, Arlington, VA, 699-708.*

In order to analyze effects on water quality of tire shreds below the water table, a field trial consisting of three sites near Orono, Maine was constructed. Each site contained 1.4 metric tons of tire shreds with steel and glass belts. The shreds, which had a maximum size of 75 mm, were placed in trenches below the water table. Each of the three sites had different soil types, including marine clay, glacial till, and peat. At each site, a water sampling well was positioned upgradient in order to analyze background water quality. Also at each site a well was put directly in the trench, and two to four wells were placed at distances of 0.6 and 3 m downgradient of the trench. Water samples were collected from the wells over a four year period and were filtered and analyzed for metals and volatile and semivolatile organic compounds. The test results show that concentrations of metals with a primary drinking water standard were not significantly affected by the presence of tire shreds. Arsenic, cadmium, and lead were all consistently below detection limits. Copper was usually below maximum levels, and background concentrations were higher than the concentrations in the trenches, suggesting that the tire shreds did not add to the levels of dissolved copper. Barium was also present, but was less than the standard maximum and was not inclined to travel downgradient. As far as metals with secondary drinking water standards were concerned, iron, manganese, and zinc decreased with time or decreased downgradient, while silver, aluminum, and sodium had similar concentrations in the trenches and in the upgradient wells. Some unfiltered samples showed the presence of iron oxide particles and an increased concentration of manganese and zinc, but these all decreased with time. Chromium had higher concentrations in the unfiltered samples in some cases as well. Concentrations of organic compounds were below method detection limits. Tire shreds may emit semivolatile organics, but concentrations decrease downgradient. Overall, this study illustrates that tire shreds beneath the water table have insignificant off-site effects on water quality.

Twin City Testing. (1990). Environmental Study of the Use of Shredded Waste Tires for Roadway Sub-grade Support. *Twin City Testing Corp., St. Paul, MN, for Waste Tire Management Unit, Site Response Section, Groundwater and Solid Waste Division, Minnesota Pollution Control Agency, St. Paul, MN.*

Field samples of soil and groundwater were taken from two existing road sites where tires were used in the subgrade. The road was built over a wetland, so part of the tire shred structure was under water. In the water samples, concentrations of barium, cadmium, chromium, and lead exceeded Recommended Allowable Limits near the tire shreds but not in the control samples, suggesting that the tire shreds could have an effect on groundwater quality. The study determined that metal concentrations in the soil samples were similar to the naturally occurring concentrations, but the report concludes that further studies with groundwater monitoring wells should be carried out. (See *Laboratory* section for summary of laboratory study.)

Laboratory

Miller, W.L. and Chadik, P.A. (1993). A study of waste tire leachability in potential disposal and usage environments: Amended volume 1: Final report. *The College of Engineering, University of Florida, Department of Environmental Engineering Sciences.*

This report consists of both laboratory and field studies in which tire shreds obtained from the Southwest Landfill in Alachua County, Florida were used. The shreds used had been stockpiles in the landfill for less than a year and were rinsed with tap water prior to testing. The 38-day laboratory studies used nine different aqueous solutions with varied conductivity and pH. Three different size-ranges of tire shreds were tested: 2-4 in., 1-2 in., and 0.5-1 in. (See *Above Water Table* section for summary of field study.)

Moo-Young, H., Sellasie, K., Zeroka, D., and Sabnis, G. (2003). Physical and Chemical Properties of Recycled Tire Shreds for Use in Construction. *Journal of Environmental Engineering*, 129, 10, 921-929.

For this study, tires were obtained from a stockpile in Wind Gap, Pennsylvania. When tire shreds were placed in water for 24 hours, the pH decreased from 7.95 to 6.98, showing a slight acidity of the tire sample, which could have been caused by exposed steel in the tire shreds. Different chemical analyses were weighed against the size of tire shreds. Overall, the total organic carbon (TOC) levels and the pH both decreased with an increase in shred size. Seven-day continuous and pause flow column tests were also conducted. With continuous flow, iron effluent, pH, turbidity, and TOC all experienced

an overall decrease with time. Pause flow, which was intended to simulate a worst case scenario in which drainage flow would be intermittent, such as in the clogging of a geotextile, caused an increase in iron, pH, turbidity, and TOC. The report suggests that an increase in iron during pause flow, which in turn led to a increase in pH, was due to rusting of the exposed steel that was caused by exposure with both water and oxygen. The increase in TOC was likely due to the presence of stagnant water in the bottom of the column for periods of the pause flow experiment. The study concludes that tire shreds could have negative environmental impacts when placed below the water table in areas where the drainage is poor.

San Miguel, G., Fowler, G.D., Sollars, C.J. (2002). The leaching of inorganic species from activated carbons produced from waste tyre rubber. *Water Research*, 36, 8, 1939-1946.

In this study, leachate from carbon was obtained from pyrolysis of waste tires. Powdered waste tire rubber was obtained from Duralay Ltd. and was heated in a furnace at 700°C with volatile compounds constantly being removed. Steam activation and carbon dioxide activation were then carried out at higher temperatures and carbons were obtained and ground to a particle size < 150 µm. It was found that the rubber contained large amounts of sulfur and zinc, as well as significant amounts of iron, calcium, magnesium, and aluminum. There were trace amounts of lead, cadmium, copper, chromium, and molybdenum. When exposed to neutral pH conditions, it was determined that the availability of the inorganic compounds was limited, but leaching of all inorganic species, except molybdenum, was found to increase with an increase in pH. Under neutral pH conditions, the only component to exceed World Health Organization drinking water standards was sulfur.

Twin City Testing. (1990). Environmental Study of the Use of Shredded Waste Tires for Roadway Sub-grade Support. *Twin City Testing Corp., St. Paul, MN, for Waste Tire Management Unit, Site Response Section, Groundwater and Solid Waste Division, Minnesota Pollution Control Agency, St. Paul, MN.*

In this report by Twin City Testing, laboratory leach tests and field tests are summarized. For the laboratory study, seven tires from a stockpile ranging in age from 15 to 20 years and seven tires from a stockpile ranging in age from 5 to 10 years were used. Tests were done at various pH levels, with a pH of 3.5 to represent a worst-case scenario. The study determined that metals were found in higher concentrations at lower pH levels. When compared to Minnesota Department of Health Recommended Allowable Limits for drinking water, some metals, including arsenic, cadmium, chromium, lead, selenium, and zinc exceeded the limits in certain situations. Higher concentrations of organics occurred

in the higher pH settings and in the newer tires. (See *Below Water Table* section for summary of field study.)

Zelibor, J.L. (1991). Leachate from Scrap Tires: RMA TCLP Report. *Education Seminar on Scrap Tire Management, Scrap Tire Management Council, Washington, D.C.*, 381-391.

This paper reports on the laboratory tests done by Radian Corporation for the Rubber Manufacturers Association (RMA). Material samples were obtained from the manufacturers of different tire components. Whole tire samples were also obtained from tire manufacturers. The samples were cut into pieces 1 cm or less, and leachate was extracted without exposure to air. Radian followed the Toxicity Characterization Leaching Procedure (TCLP) and Extraction Procedure (EP) Toxicity procedures on cured, uncured, ground, and unground rubber products. Gas chromatography, mass spectrometry, and atomic absorption spectrometry were used to detect the target chemicals. From the resulting levels of detection for various volatiles and semi-volatiles, the report states that Radian's findings provide "strong evidence that tire shreds pose no threat to groundwater," but it also suggests that field trials should be carried out.

Asphalt Pavement

Azizian, M.F., Nelson, P.O., Thayumanavan, P., and Williamson, K.J. (2003). Environmental impact of highway construction and repair materials on surface and ground waters: Case study. Crumb rubber asphalt concrete. *Waste Management*, 23, 8, 719-728.

The possible environmental effects of crumb rubber asphalt concrete were considered in this study. Several laboratory chemical and toxicity tests were performed on rubber modified asphalt. Toxicity tests were carried out using algae and water fleas. In batch leaching tests, in which the material was reduced to 1/4-in. in size, it was seen that concentrations of aluminum and mercury after only 10 hrs were about 50% of the final concentrations reached after several days. The likely source of the mercury was the asphalt binder, while the likely source of the aluminum was the crumb rubber modified asphalt. Benzothiazole and 2(3H)-benzothiazolone were the organic compounds in the leachate with the highest concentration. Flat plate leaching tests were also conducted, but limited results were presented.

The leachate exhibited some toxic effects to *S. capricornutum* and *D. magna*. Soil sorption tests showed that soils were able to render the leachate nontoxic. The results

also found that benzothiazole is a volatile and biodegradable compound. The authors concluded that contaminants from rubber modified asphalt will be degraded or retarded by natural processes and should not be transported into nearby soils and groundwater.

Vashisth, P., Lee, K.W., and Wright, R.M. (1998). Assessment of water pollutants from asphalt pavement containing recycled rubber in Rhode Island. *Transportation Research Record* 1626, 95-104.

In this three-phase laboratory experiment, crumb rubber modifier (CRM) contaminants were first identified. Secondly, a water quality evaluation was performed on CRM samples under simulated environmental conditions. Similarly, water quality evaluations were performed on samples exposed to simulated rainfall conditions.

In the first phase of the experiment, samples of CRM were brought to a pH of 2 using nitric acid and were analyzed for trace metals using an atomic absorption spectrophotometer. CRM produced by both wet or dry processes were tested. It was found that zinc concentrations were three to four orders of magnitude greater than the concentrations of the five other trace metals that were tested for (cadmium, chromium, copper, lead, and nickel). The most likely source of the zinc was thought to be zinc oxide, an ingredient in CRM. Leachate from CRM was also tested for selected organic compounds. Four compounds were found in one or more of the samples, but no polycyclic aromatic hydrocarbons were found.

In the second part of the study HMA samples were prepared by the Marshall method. Control specimens without CRM, as well specimens with CRM asphalt specimens prepared by the wet and dry processes. Samples were tested at pH levels of 2.0, 7.0, and 12.0 and at temperatures of 20°C and 45°C to represent the average and maximum asphalt pavement temperatures. Six metals were included in the analysis: cadmium, chromium, copper, lead, nickel, and zinc. There was a general trend that higher concentrations of these metals were released from the wet and dry process CRM samples, but for cadmium, chromium, and copper the difference was small. For lead nickel, and zinc the differences were generally larger for the dry process samples than the wet process samples possibly because a greater amount of CRM is used in the dry process. Higher concentrations of benzothiazole were found wet and dry process CRM samples compared to the control samples. The sum of the concentrations of the 24 polycyclic aromatic hydrocarbons included in the analysis ranged from 60.2 to 601 ng/L for control samples, 374 to 710 ng/L for dry process CRM samples, and 97.3 to 727 ng/L for wet process CRM samples. The upper limit of the range for each sample category is similar.

For the third part of the study, compacted samples were exposed to light, traffic, and rain simulations. Wet, dry, and chip seal samples were tested. For the chip seal control, the samples were treated with an emulsified asphalt binder, while an asphalt rubber binder was used along with the wet process for a comparison. Rather than using simulated light,

the samples were actually placed in an HDPE box with a clear lid and were exposed to natural sunlight. To simulate traffic, a steel wheel mobile compactor was used on top of Teflon plates. A 125mm, 30 min. rainstorm was simulated using water with pH values of 4.3 and 7.0 for different samples. Metal concentrations were low for both pH values, and only two metals were detected for the dry process samples. It was determined that there was little variation in metal concentrations between the wet, chip seal, and control samples. Runoff from the wet and dry process CRM specimens had concentrations of benzothiazoles, hydroxybenzotriazole, and PAH's that were higher than the control case.

Toxicity

Abernethy, S. (1994). The acute lethality to rainbow trout of water contaminated by an automobile tire. *Ontario Ministry of Environment and Energy*, 22 p.

In this laboratory study, rainbow trout fry, daphnia magna, ceriodaphnia dubia, and fathead minnows were exposed to tire-contaminated water to determine acute toxicity. Water samples were also tested for a variety of inorganic and organic compounds. To prepare a water sample, a 6.9 kg used radial automobile tire was first soaked and washed with water and then submerged in a water bath. A tire-free water sample was established as a control. It was found that all of the tire-contaminated water samples were lethal to the trout, but not the other organisms. Chemical analyses showed that of the metals tested for, only zinc was found in higher concentrations in the tire water in comparison to the control, but zinc concentrations were still below lethal levels. Of the 143 organics tested for, 4 were detected in all of the tire water samples – aniline, 4-(1-methyl-1-phenylethyl)-phenol, benzothiazole, and 4-(2-benzothiazolythio)-morpholine. The study was not able to determine what chemicals were toxic to the rainbow trout but suggested that phenolic, nitrogen aromatic, and heterocyclic compounds be studied further. It was determined that the toxic chemicals were most likely part of the rubber manufacturing process and were leached directly from the rubber material, as opposed to from the tire surface.

Birkholz, D.A., Belton, K.L., and Guidotti, T.L. (2003). Toxicological Evaluation for the Hazard Assessment of Tire Crumb for Use in Public Playgrounds. *Journal of the Air & Waste Management Association*, 53, 7, 903-907.

Laboratory tests were performed on tire crumb to determine any environmental and health risks of using the shredded tire material as a ground cover at playgrounds. To determine the toxicity of tire leachate to aquatic organisms, analysis was done on species of luminescent bacteria, invertebrates, fish and algae. For each test, 250 g samples of tire

crumb were leached in 1 L of water, which was then filtered to eliminate particles. Using standard testing methods it was determined that all of the samples were toxic to all four of the species tested. However, it was also determined that the toxicity was 59% less in leachates from tire crumb that had been on playgrounds for three months. More laboratory tests showed that introduction of nutrients and sewage seed and constant air exposure for five days provided 73-86% reductions in toxicity. Overall, it was determined that leachate from fresh tire crumb could have moderate effects if runoff is not diluted, but in most environmental scenarios, enough dilution and natural processes would be present to decrease toxicity enough for there to be very little risk of surface water or groundwater contamination.

Christiansson, M., Stenberg, B., and Holst, O. (2000). Toxic Additives – A Problem for Microbial Waste Rubber Desulphurisation. *Resource and Environmental Biotechnology*, 3, 1, 11-21.

The purpose of this study was to look at the effects that compounds added to tire rubber have on microbial desulphurization. Various microorganisms were exposed to different chemicals used in tire rubber. Microorganism growth was monitored to determine if the chemicals could inhibit growth and in turn affect the sulfur oxidation performance of the microorganisms. It was found that certain accelerators and vulcanizing agents did inhibit growth of some of the tested microorganisms. Samples of tire rubber were also leached using five different solvents. Microorganisms behaved differently depending on the solvent used and experienced more growth in the leachate obtained with an organic solvent. More tests would have to be conducted to determine what concentrations of the chemicals initiated inhibited growth in order to determine limiting concentrations for use in tire rubber.

Day, K.E., Holtze, K.E., Metcalfe-Smith, J.L., Bishop, C.T., and Dutka, B.J. (1993). Toxicity of Leachate from Automobile tires to Aquatic Biota. *Chemosphere*, 27, 4, 665-675.

A laboratory study was performed in order to determine if harmful chemicals are leached from tires submerged in fresh water. New tires and road-worn tires, as well as tires from a 10-year-old floating breakwater were tested for toxicity to certain aquatic organisms, as well as for volatility of any leached chemicals. For each test a whole tire was put into 300 L of natural groundwater. On days 5, 10, 20, and 40, 40 L of water was removed from each sample for testing. It was determined that the tires obtained from the floating breakwater did not leach any chemicals deemed toxic to the species tested. Both the new and used tires were lethal to rainbow trout, the used tires being more toxic than the new ones. The water continued to be toxic even after the tires were removed, signifying that

chemicals present were slow to degrade and were not volatile. The chemicals leached were not harmful to cladocerans or fathead minnows.

Evans, J.J., Shoemaker, C.A., and Klesius, P.H. (2000). In vivo and in vitro effects of benzothiazole on sheepshead minnow (*Cyprinodon variegatus*). *Marine Environmental Research*, 50, 1-5, 257-261.

The purpose of this laboratory study was to determine if benzothiazole is the component in tire leachate that is toxic to the nervous systems of larval sheepshead minnow. Sheepshead minnow larvae were exposed to different concentrations of benzothiazole, ranging from 3.75 to 60 mg/L. After 5 days, death occurred in all minnow exposed to 60 mg/L. At each concentration there was a significant decrease in larval growth and gill cells were affected. Exposed larvae were found to weigh substantially less than control larvae. No lesions were observed in the eyes or brains of the larvae. The results of the in vivo and in vitro experiments suggest that benzothiazole is not a neurotoxicant, but it does cause fatal respiratory problems in sheepshead minnow larvae.

Johnson, B.L., Belluck, D.A., and Melby, A.M. Hazard Analysis and Risk Management of Road Subbase Materials Using the Comparative Risk Bioassay Methodology. *Mn/DOT*.

Mn/DOT did a comparative study on the environmental toxicology of wood chips and tire shreds for use as lightweight fill material. A goal of the study was to determine if tire shreds could replace wood chips in submerged road bases. Chemicals were extracted from wood chip and tire shred samples over a 7-day period. Water flea, minnow, algae, lettuce, and earthworm specimens were exposed to various concentrations of the two leachates. Chemical tests were also performed on the leachates. Acute and chronic toxicity were effects of the wood and tire leachates in the water flea, minnow, and algae specimens. It was found that the tire leachate contained concentrations of zinc that exceeded the established Health Risk Level, while the wood leachate contained 2-methylphenol concentrations that exceeded this level. Overall, it was established by Mn/DOT that the toxicity of the tire shred leachate was greater than that of the wood chips, thus making the substitution of wood chips with tire shreds for submerged lightweight fill not viable at the time of the study.

Stephensen, E., Adolfsson-Erici, M., Celander, M., Hulander, M., Parkkonen, J., Hegelund, T., Sturve, J., Hasselberg, L., Bengtsson, M., and Forlin, L. (2003) Biomarker

responses and chemical analyses in fish indicate leakage of polycyclic aromatic hydrocarbons and other compounds from car tire rubber. *Environmental Toxicology and Chemistry*, 22, 12, 2926-2931.

The sublethal effects of tire rubber on fish were explored in this toxicity study. Two types of new tires were used – one with highly aromatic (HA) oil-containing tread and one with no HA oil in the tread, but HA oil in the other parts of the tire. Each tire was added to a 400 L fish tank with a flow rate that allowed for complete water turnover every 24 hrs. Male and female juvenile rainbow trout were added to the tanks 24 hrs after the addition of the tires. A control setup was also implemented using a fish tank with no tire present. Samples of fish were removed after 1 day from the tanks and killed and again after 2 weeks for tests to be performed. It was found that fish exposed to the tires had a higher liver somatic index, but the index was higher in fish exposed to the tires with no HA oil in the treads than in fish exposed to the tires with HA oil in the treads. After 1 day, cytochrome P4501A1 (CYP1A1) levels in the fish exposed to either type of tire were at least 15 times higher than in the control fish. After 2 weeks however, the CYP1A1 levels in the fish exposed to HA-free tread tires were only three times higher than the control, while the fish exposed to HA tread tires had levels 13 times higher than the control. Increases in glutathione S-transferase, glutathione reductase, and glucose-6-phosphate were observed in fish exposed to tires, but were higher in fish exposed to HA oil-free tread tires. Hydroxylated polycyclic aromatic hydrocarbons and aromatic nitrogen compounds were found in the bile of fish exposed to tires. In another portion of the experiment, fish were injected with either HA oil, mild extracted solvate (MES) oil, or β -naphthoflavone. The fish were kept in tanks and then sampled after five days. CYP1A1 levels increased in the fish injected with HA oil, but not mild extracted solvate. The results show that fish exposed to HA oils uptake undesirable compounds leached from the tires. Tires that contain no HA oils in the treads still leach the oils from other parts of the tire. The study recommends that HA oils be replaced with MES oils in tire manufacturing.

Leachate Collection

Aydilek, A.H., Madden, E.T., and Demirkan, M.M. (2006). Field evaluation of a leachate collection system constructed with scrap tires. *Journal of Geotechnical and Geoenvironmental Engineering*, 132, 8, 990-1000.

In this study, a field test was constructed consisting of two leachate collection cells at a landfill in Oakland, MD. Cell 1, the control cell, was built using a 0.6 m limestone gravel layer for leachate collection. Cell 2 consisted of a geosynthetic clay liner, a 1.5 mm textured HDPE geomembrane, and a geocomposite drainage layer, topped with tire chips. A 0.25 m layer of wire-free tire chips was placed directly on the geocomposite layer, and a 0.75 m layer of tire chips with wire was placed on top. Leachate samples were

collected from the cells every 3-4 months over the 4-year monitoring program. It was found that the pH levels for Cells 1 and 2 were generally similar and fell between 6 and 8, suggesting that the tire chips did not leach alkaline or acidic compounds. In addition, the electrical conductivity of Cell 2 was lower than that of Cell 1. It was seen that concentrations of iron and zinc, as well as copper, chromium, and barium to a lesser extent, were higher in the tire chip leachate. The only inorganics to exceed USEPA maximum concentration limits and water quality limits were iron and manganese, but the concentrations were also high in the control cell, suggesting that those metals leached from the municipal waste in the landfill. This was also the case for some volatile organic compounds. The study concluded that tire chips can be used for a landfill leachate collection system, but the tire chips may not be appropriate for use near drinking water sources.

Leach Fields for On-Site Waste Water Treatment Systems

Miller, W.L. and Chadik, P.A. (1993). A study of waste tire leachability in potential disposal and usage environments: Amended volume 1: Final report. *The College of Engineering, University of Florida, Department of Environmental Engineering Sciences.*

In the field test, septic drainage field leachate from a 1 to 2-in. tire shred-filled trench was compared with a gravel-filled control trench. Each trench's leachate was collected in two locations with landscaping fabric-wrapped PVC pipe connected to collection sumps. Measurements were taken for about 100 days. Samples were tested for metals, semi-volatile organics, and volatile organics. It is impossible to derive meaningful conclusions from the semivolatile results because of the presence of analytes of interest in the trip and laboratory blanks. Five volatile compounds were detected in the wastewater input to the system. For these compounds the concentration for samples taken from the tire shred filled trench was less than or about the same as in the gravel filled trench. Two compounds (135-TMB and MIBK) were found in samples from the tire shred filled trenches, but the concentrations were less than 7 µg/L. Arsenic and cadmium was present, but the concentrations from the tire shred and gravel trench about the same or lower than the distribution box suggesting that the source was the influent.

Sengupta, S., and Miller, H. (2004), "An Evaluation of Recycled Tire Shreds as a Substitute for Gravel in Residential Soil Absorption Systems," Civil Engineering Practice – Journal of the Boston Society of Civil Engineers, Vol. 19, No. 1, pp. 33-52.

A laboratory and field study was conducted of the use of tire shreds as a substitute for gravel in residential septic tank leach fields. The laboratory portion of the study included batch and column leaching tests. The mean concentrations of chromium and copper, which have primary drinking water standards, were below the applicable regulatory limits. The tire shreds did leach iron, manganese, aluminum, and zinc, as well as, low levels of chloride and sulfate. In the field portion of the study, there were three leach field trenches. One contained conventional gravel while the other two contained tire shreds with a maximum size of about 2 in. (50 mm). Samples were collected from pan lysimeters placed beneath each trench as well as samples of the influent taken from the distribution box. Samples were taken biweekly over a seven month period. The results showed that the effluent quality parameters (biological oxygen demand (BOD), fecal coliform (FC), ammonium-nitrogen ($\text{NH}_4^+\text{-N}$), and nitrate-nitrogen ($\text{NO}_3^-\text{-N}$)) from the tire shred filled trenches were the same as for the conventional gravel filled trench. However, the tire shred filled trench required 30 to 60 days longer to establish a stable biomat. The tire shred filled trenches did not leach toxic substances at concentrations above their regulatory allowable limits, except for manganese. However, the manganese concentrations were the same for the tire shred and gravel filled trenches. The authors concluded that tire shreds are an acceptable alternative to conventional gravel.